

Appendix D Species CE Conceptual Models

Table of Contents

APPENDIX D	SPECIES CE CONCEPTUAL MODELS.....	1
INTRODUCTION.....		6
Species Characterization and Conceptual Models.....		6
References		9
SPECIES CONCEPTUAL MODELS.....		11
MAMMALS		12
D-1	Bighorn Sheep (<i>Ovis canadensis</i>)	12
D-1.1	Classification Comments	12
D-1.2	Biology and Distribution Status	13
D-1.3	Habitat and Ecology	14
D-1.4	Change Agent (CA) Characterization	17
D-1.5	Ecological Status: Key Ecological Attributes and Indicators	19
D-1.6	Conceptual Model Diagram	23
D-1.7	References for the CE	23
D-2	Black-tailed Prairie Dog (<i>Cynomys ludovicianus</i>).....	30
D-2.2	Biology and Distribution Status	30
D-2.3	Habitat and Ecology	35
D-2.4	Change Agent Characterization	38
D-2.5	Ecological Status: Key Ecological Attributes and Indicators	40
D-2.6	Conceptual Model Diagram	42
D-2.7	References for the CE	42
D-3	Coues White-tail Deer (<i>Odocoileus virginianus couesi</i>)	47
D-3.1	Taxonomic or Classification Comments.....	47
D-3.2	Biology and Distribution Status	49
D-3.3	Habitat and Ecology	50
D-3.4	Change Agent Characteristics	53
D-3.5	Ecological Status: Key Ecological Attributes and Indicators	54
D-3.6	Conceptual Model Diagram	57
D-3.7	References for the CE	57
D-4	Pronghorn (<i>Antilocapra americana</i>).....	62
D-4.1	Classification Comments	62
D-4.2	Biology and Distribution Status	63
D-4.3	Habitat and Ecology	66
D-4.4	Change Agent Characterization	68

D-4.5	Ecological Status: Key Ecological Attributes and Indicators	71
D-4.6	Conceptual Model Diagram	75
D-4.7	References for the CE	75
REPTILES.....		81
D-5	Desert Box Turtle (<i>Terrapene ornata</i> subspecies <i>luteola</i>)	81
D-5.1	Classification Comments	81
D-5.2	Protection Status	82
D-5.3	Biology and Distribution Status	82
D-5.4	Habitat and Ecology	83
D-5.5	Change Agent Characteristics	84
D-5.6	Ecological Status: Key Ecological Attributes and Indicators	85
D-5.7	Conceptual Model Diagram	87
D-5.8	References for the CE	87
AMPHIBIANS		90
D-6	Chiricahua Leopard Frog (<i>Lithobates chiricahuensis</i>)	90
D-6.1	Classification Comments	90
D-6.2	Protection Status	91
D-6.3	Biology and Distribution Status	91
D-6.4	Habitat and Ecology	92
D-6.5	Change Agent Characteristics	95
D-6.6	Ecological Status: Key Ecological Attributes and Indicators	100
D-6.7	Conceptual Model Diagram	104
D-6.8	References for the CE	104
ASSEMBLAGES.....		112
D-7	Grassland Birds	112
D-7.1	Introduction	112
D-7.2	Individual Bird Highlights	113
D-7.3	Change Agent Characteristics	120
D-7.4	Summary	124
D-7.5	Ecological Status: Key Ecological Attributes and Indicators	124
D-7.6	Conceptual Model Diagram	128
D-7.7	References for the CE	128
D-8	Nectivorous Bats: Lesser long-nosed bat (<i>Leptonycteris yerbabuenae</i>), Mexican long-nosed bat (<i>Leptonycteris nivalis</i>), Mexican long-tongued bat (<i>Choeronycteris mexicana</i>)	134
D-8.1	Taxonomic or Classification Comments	134
D-8.2	Individual Bat Highlights	134

D-8.3	Summary	140
D-8.4	Change Agent Characterization	141
D-8.5	Ecological Status: Key Ecological Attributes and Indicators	143
D-8.6	Conceptual Model Diagram	145
D-8.7	References for the CE	145

Tables

Table D-1. Species conservation elements (species CEs) selected for the Madrean Archipelago REA.6	
Table D-2. Key Ecological Attributes (KEA) used to determine the ecological status of desert bighorn sheep (<i>Ovis canadensis</i>).....	20
Table D-3. Key Ecological Attributes (KEA) used to determine the ecological status of Black-tailed Prairie Dog (<i>Cynomys ludovicianus</i>)	41
Table D-4. Key Ecological Attributes (KEA) used to determine the ecological status of Coues white-tailed deer (<i>Odocoileus virginianus couesi</i>)	55
Table D-5. Key Ecological Attributes (KEA) used to determine the ecological status of North American Pronghorn (<i>Antilocapra americana Mexicana</i> and <i>A. a. Americana</i>)	72
Table D-6. Key Ecological Attributes (KEA) related to the ecological status of the Desert Box Turtle (<i>Terrapene ornata luteola</i>)	86
Table D-7. Key Ecological Attributes (KEA) related to the ecological status of the Chiricahua Leopard Frog (CLF) (<i>Lithobates chiricahuensis</i>)	101
Table D-8. Examples of habitat characteristics for grassland bird species.....	118
Table D-9. Key Ecological Attributes (KEA) related to the ecological status of the grassland bird assemblage	126
Table D-10. Key Ecological Attributes (KEA) used to determine the ecological status of nectar feeding bats: Lesser long-nosed bat (<i>Leptonycteris yerbabuenae</i>), Mexican long-nosed bat (<i>Leptonycteris nivalis</i>), and Mexican long-tongued bat (<i>Choeronycteris mexicana</i>).....	144

Figures

Figure D-1. Desert bighorn sheep (<i>Ovis canadensis</i>) in southwestern New Mexico	13
Figure D-2. Conceptual model for the desert bighorn sheep, showing key ecological attributes	23
Figure D-3. Black-tailed prairie dog (<i>Cynomys ludovicianus</i>).	30
Figure D-4. Conceptual model for the black-tailed prairie dog, showing key ecological attributes ..	42
Figure D-5. Coues white-tail deer (<i>Odocoileus virginianus couesi</i>).	48
Figure D-6. Conceptual model for the North American Coues white-tailed deer, showing key ecological attributes	57
Figure D-7. American pronghorn (<i>A. a. americana</i>), Animas Valley, Arizona.....	63
Figure D-8. American pronghorn (<i>A. a. americana</i>), Las Cienegas National Conservation Area, Arizona.....	63
Figure D-9. Conceptual model for the North American Pronghorn, showing key ecological attributes	75
Figure D- 10. Desert Box Turtle (<i>Terrapene ornata luteola</i>), near Patagonia, Arizona.	81
Figure D- 11. Conceptual model for the Desert Box Turtle, showing key ecological attributes.....	87
Figure D-12. <i>Chiricahua leopard frog</i> (<i>Lithobates chiricahuensis</i>) - <i>Sycamore and Scotia Canyons</i> , Arizona.	90
Figure D-13. Conceptual model for the Chiricahua Leopard Frog, showing key ecological attributes	104
Figure D-14. Conceptual model for the grassland bird assemblage, showing key ecological attributes	128
Figure D-15. Conceptual model for nectar feeding bats, showing key ecological attributes.....	145

Introduction

Species Characterization and Conceptual Models

The conceptual models combine text, concept diagrams, and tabular summaries in order to state assumptions about the ecological composition, structure, dynamic processes, and interactions with major CAs within the ecoregion. These conceptual models lead then to spatial models to enable gauging the relative ecological status of each Conservation Element (CE), which will be completed in a later task of the REA. Below is described the content included for each species CE. The MAR is a highly biologically diverse region, but only a select subset of eight species conservation elements (Table D-1) were chosen for the REA; methods for selection are described in Harkness et al. (2013). The descriptive material builds upon any existing species descriptions compiled by NatureServe (see <http://www.natureserve.org/explorer/index.htm> to search and download existing descriptions). The information developed is generally intended to cover the full range of distribution of the CE, which can extend beyond the ecoregion, but does focus on the characteristics or dynamics as they occur within this ecoregion. For this REA, additional material was added as available for each species CE, especially focused on characteristics specific to the species within the MAR.

Table D-1. Species conservation elements (species CEs) selected for the Madrean Archipelago REA.

Mammals
Bighorn Sheep (<i>Ovis canadensis</i>) Black-tailed Prairie Dog (<i>Cynomys ludovicianus</i>) Coues White-tail Deer (<i>Odocoileus virginianus couesi</i>) Pronghorn (<i>Antilocapra americana</i>)
Reptiles
Desert Box Turtle (<i>Terrapene ornata</i> subspecies <i>luteola</i>)
Amphibians
Chiricahua Leopard Frog (<i>Lithobates chiricahuensis</i>)
Assemblages
Grassland Birds Botteri's sparrow (<i>Peucaea botterii</i>) Grasshopper sparrow (<i>Ammodramus savannarum</i>) Scaled quail (<i>Callipepla squamata</i>) Baird's sparrow (<i>Ammodramus bairdii</i>) Nectivorous Bats Lesser long-nosed bat (<i>Leptonycteris yerbabuenae</i>) Mexican long-nosed bat (<i>Leptonycteris nivalis</i>) Mexican long-tongued bat (<i>Choeronycteris mexicana</i>)

Conservation Element Characterization

This section of the conceptual model includes a narrative of the CE classification, distribution, protection status, biology and distribution status, and habitat and ecology with supporting literature cited.

Classification – clarification of element taxonomy and clarification of any anomalies or changes concerning taxonomic distinctness of the species.

Distribution – Total geographic range-wide extent of the species including breeding/non-breeding or seasonal ranges if specified and with a discussion of any MAR specific range issues.

Population – Estimate of the total number of locations where the species is known to occur across its range including information on how the estimate was derived. When available MAR specific population information is provided.

Reproduction – discussion of reproduction of the species across its range including information on clutch/litter size and frequency, gestation/incubation period, seasonal timing of reproductive activities, nature and period of any parental care, age of sexual maturity, and size and general nature of breeding aggregations.

Habitat and Ecology – Description of habitats and microhabitats commonly used by the species throughout its range, including any daily seasonal and geographic variation within habitat use including considerations of mobility and connectivity between populations. Summary of the ecology of the species across its range, including information on food, phenology, predators, competitors, demographics, and disease and parasites and any other significant ecological factors.

Change Agent Effects on the CE

In this section the primary change agents and current knowledge of their effects on the CE are characterized. Some CAs have specific effects on each CE such as the alteration or destruction of habitat or disruption of connectivity between populations. This section lists the known change agents and then moves into describing the altered ecological dynamics of the CE, with a narrative on the effects of CAs on the individual CE.

Diagrams for the Model

Each species is represented by one diagram that includes key ecological attributes and stressors and change agents that are currently acting upon the key attributes. Diagrams are a visual representation of how change agents interact with key ecological attributes for each species. They are designed to highlight key drivers for species ecological health.

Ecological Status: Key Ecological Attributes and Indicators

NatureServe's ecological integrity assessment framework sets up practical criteria for assessing the ecological status of each CE within an ecoregion (Faber-Langendoen et al. 2006, Unnasch et al. 2009). This section of the conceptual model addresses Key Ecological Attributes and their potential indicators. The ecological status is a way of describing current status via criteria, functionality, or levels of attributes and asks if they are within the normal range of variation. Is it within its "proper functioning condition"? Attributes are direct and indirect measures of ecosystem status or function. Key Ecological Attributes (or their indicators) should be measured to take the "pulse" of an ecosystem or a species habitat or populations. High scores indicate high ecological integrity and high ecological functionality.

Key Ecological Attributes

The key ecological attributes for the CE within the Madrean Archipelago ecoregion are identified in this section. A **key ecological attribute** of a focal ecological resource is a characteristic of the resource's biology, ecology, or physical environment that is critical to the resource's persistence in the face of both natural and human-caused disturbance, e.g., resistance or resilience (Holling 1973, De Leo and Levin 1997, Parrish et al. 2003, Unnasch et al. 2009). Alteration of such a characteristic beyond some critical range of variation will lead to the degradation or loss of the resource within decades or less.

For each CE, a table provides identified key ecological attributes, with a brief definition, a rationale for why it is important for the CE, and a listing of stressors or change agents that might be affecting the key attribute.

Key ecological attributes of a resource include critical or dominant characteristics of the resource, such as specific characteristics of:

- a) demographic or taxonomic composition;
- b) functional composition;
- c) spatial structure;
- d) range or extent.

They also include critical biological and ecological processes and characteristics of the environment that:

- a) limit the regional or local spatial distribution of the resource;
- b) exert pivotal causal influence on other characteristics;
- c) drive temporal variation in the resource's structure, composition, and distribution;
- d) contribute significantly to the ability of the resource to resist change in the face of environmental disturbances or to recover following a disturbance; or
- e) determine the sensitivity of the resource to human impacts.

Conservation of key ecological attributes contributes to current ecological integrity and to the resilience of species in the face of large-scale or long-term stressors (Parrish et al. 2003). The ecological integrity assessment framework (Unnasch et al. 2009) identifies four classes of key ecological attributes, concerning: landscape context; resource size or extent; biotic condition; and abiotic condition. These four may overlap, and provide a guide for considering and identifying key ecological attributes. They also provide a basis for integrating information on key ecological attributes.

- "Landscape context" refers both to the spatial structure (spatial patterning and connectivity) of the landscape within which the focal resource occurs; and to critical processes and environmental features that affect the focal ecological resource from beyond its immediate geographic scope.
- "Size" refers to the numerical size and/or geographic extent of a focal resource.
- "Biotic condition" refers to biological composition, reproduction and health, and succession; and critical ecological processes affecting biological structure, functional organization (e.g., food-web guild structure), and interactions.
- "Abiotic condition" refers to physical environmental features and dynamics within the geographic scope of the focal resource that significantly shape biotic conditions, such as fire, weather, and hydrologic regimes; and soil and geological conditions and dynamics.

Taken together these attributes tell the story of the current status of a species or an ecosystem.

Indicators of Key Attributes

Assessing the status of key ecological attributes requires explicit identification of indicators (also called metrics) – specific means for measuring their status. These are the detailed metrics that measure the amount or status of each key attribute. There are many potential indicators, and the choice is largely dependent on the purpose of the assessment and available data. An indicator may be a specific, measurable characteristic of the key ecological attribute; or a collection of such characteristics combined into a "multi-metric" index. Such indicators directly evaluate the condition of the KEAs and their responses to stressors (change agents).

Alternatively, indicators may evaluate the severity and extent of the stressors themselves. Such "Stressor" indicators may consist of a single measurement type, or a collection of such measurements

combined into a multi-metric stressor index. Indicators of stressors are often used as indirect indicators of a key ecological attribute, because data on stressor condition is often far more readily available than data on direct indicators. Examples of stressor-based indicators include measures of overall landscape development such as the Landscape Condition Model methodology (Comer and Hak 2009, Comer and Faber-Langendoen 2013); measurements of invasive non-native annual grass distributions that affect fire regimes; or measurements of fragmentation due to development.

Once the REA for the MAR has moved into actual analysis and assessment of status, the indicators used for each KEA for each CE will be identified and explained in the CE conceptual models, along with results of the status assessment. For now, no indicators are listed.

References for the CE

Literature is listed that is relevant to the classification, distribution, floristic composition, ecological processes, threats, stressors, or management of the CE, in some cases from portions of its range outside of the ecoregion. These are not exhaustive literature surveys, rather are an accumulation of known references. Some documents may be listed that are not cited in the narrative text.

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Species Conceptual Models

Mammals

D-1 Bighorn Sheep (*Ovis canadensis*)

D-1.1 Classification Comments

Ancestors of the Desert Bighorn Sheep, also referred to as Mountain Sheep or *Borrego cimarrón*, in the Bovidae family, originated in the mountain and desert regions of Eurasia and crossed the Bering land bridge in the late Pleistocene (Boyce et al 1999, San Diego Zoo Global Library 2009). This old world sheep evolved into many different and distinct wild sheep (Figure D-1), each adapted to their native habitats.

The Integrated Taxonomic Information System (www.itis.gov; accessed 23 March 2013) recognizes seven subspecies of bighorn sheep: an extinct subspecies of American bighorn (*Ovis Canadensis auduboni*, Merriam, 1901); California bighorn sheep (*O. c. californiana*, Douglas, 1829); Rocky Mountain bighorn sheep (*O. c. Canadensis*, Shaw, 1804); Peninsular desert bighorn sheep (*O. c. cremnobates*, Elliot 1904); Desert bighorn sheep (*O. c. Mexicana*, Merriam, 1901); Nelson's desert bighorn sheep (*O. c. nelsoni*; Merriam, 1897); and the Weems desert bighorn sheep (*O. c. weemsi*, Goldman, 1937)(ITIS, accessed 23 March 2013; Boyce et al 1999).

There has been some disagreement regarding the number of subspecies and the validity of some population distinctions. Wehausen and Ramey (2000) concluded that the extinct American bighorn and populations of California bighorn occurring in British Columbia and Washington are genetically grouped with Rocky Mountain bighorn; extirpated native populations of California bighorn in northeastern California, Oregon, northern Nevada, and southwestern Idaho are actually Nelson's bighorn; and the California bighorn is restricted to just the central and southern Sierra Nevada in California. Wehausen and Ramey (2000) also suggested that the California and Rocky Mountain bighorn Sierra Nevada population in Yakima County, Washington have a unique mtDNA haplotype (Wehausen et al. 2005; USFWS 2008) and should be recognized as a new distinct subspecies (*Ovis canadensis sierrae*).

Ramey (1995) examined mtDNA variation in Rocky Mountain bighorn populations in the southwestern United States and adjacent northern Baja California and found a lack of similar traits between mtDNA haplotype distributions to justify many of the current subspecies designations. Additionally, Ramey (1993) and Wehausen and Ramey (1994) found no morphological support for the separation of desert sheep into four subspecies. Ramey (1995) suggested that Nelson's, Mexicana, Peninsular, and Weems desert sheep populations probably should be recognized as a single polytypic subspecies (Nelson's desert bighorn sheep) on the basis of morphological and mtDNA data. As such, the subspecies would include bighorn sheep populations in the Peninsular Ranges of southern California and Baja California. A study of nuclear DNA markers is needed to test this taxonomic arrangement (Ramey 1995). The genetic integrity of these taxonomic subspecies could potentially be complicated by translocation and reintroduction efforts to bring back declining or extirpated populations in the region; bighorn sheep reintroduction efforts began in Arizona in 1979 (McKinney et al 2006).

Figure D-1. Desert bighorn sheep (*Ovis canadensis*) in southwestern New Mexico , courtesy of New Mexico Game and Fish Department website <http://www.wildlife.state.nm.us/conservation/bighorn>



The focus for the Madrean Ecoregional assessment is on Desert bighorn sheep (*O. c. Mexicana*, Figure D-1), unless otherwise noted.

D-1.2 Biology and Distribution Status

D-1.2.1 Distribution

North America: The bighorn sheep historically ranged across the majority of the mountainous western United States, extending from British Columbia to southern Baja California and northwestern Sonora and Chihuahua (Mexico), and as far east as portions of North Dakota, South Dakota and Nebraska and parts of western Texas.

Madrean Archipelago ecoregion: The Mexicana desert bighorn sheep (*O.c. Mexicana*) is the only subspecies historically distributed throughout the majority of Arizona, New Mexico and northern Mexico (Cowan 1940, Hall 1981). Today bighorn are reduced to fragments of their previous range, with remaining populations restricted to small isolated mountain and canyon areas. Within the Madrean Archipelago ecoregion, occupied and unoccupied bighorn habitat includes the Peloncillo Mountains north of Interstate 10 at San Simon, AZ and extending into New Mexico; the Galiuro Mountains and Galiuro Wilderness encompassing Redfield Canyon Wilderness and Kelsey Canyon; the Galiuro Mountains encompassing Table Mountain and Arivaipa Canyon northeast of Mammoth, AZ; the Superstition Mountains and Superstition Wilderness near the Salt River north of Superior, AZ; Pusch Ridge Wilderness in the Santa Catalina Mountains; the Silverbell Mountains and Ironwood Forest National Monument west of Tucson, AZ; the Hatchet Mountains of New Mexico; and New Mexico's Red Rock area of the Gila River (AZGFD 2013, NMDGF 2013).

D-1.2.2 Population

In 1991, the total population of desert bighorn in North America was estimated to be 23,055 (Valdez and Krausman 1999). In 1992, the Endangered Species Technical Bulletin 17(3-8):6 reported that Peninsular Ranges desert bighorn populations in the U.S. declined to 1,171 by 1979 and to fewer than 400 in the early 1990s. Helicopter based surveys of the principle mountain ranges in Sonoran in 1992 produced an

estimate of 2,000 wild sheep for Sonoran (ICUN). Desert bighorn populations occurring in Mexico have CITES Appendix II status (CITES 2010; www.itis.gov, accessed 23 March 2013). California bighorn occurring in Sierra Nevada, CA and the Rocky Mountain bighorn in peninsular ranges of California are listed as Endangered in the U.S. The IUCN lists Weems desert bighorn as Critically Endangered, Peninsular desert bighorn as Endangered, and the remaining subspecies as Lower Risk. New Mexico successfully delisted the desert bighorn from the state's Threatened and Endangered species list in 2011 (NMDGF 2013).

D-1.2.3 Reproduction

Breeding Systems: During mating season mature males compete for access to females through vigorous head butting behavior. During the rest of the year they live amiably in small bachelor herds apart from female herds.

Gestation: Gestation lasts about 175 days (Geist 1971, Shackleton et al. 1999).

Litter size: Litter size is 1, twins occurring rarely (Geist 1971, Turner and Hansen 1980).

Breeding Season: The timing of the mating season varies throughout their range. Bighorns in southwestern Arizona deserts have an extended season encompassing several months (Krausman et al. 1999), but the season is relatively later and shorter elsewhere, generally November in the northern part of the range (Shackleton et al. 1999), November-December in some southern California mountains (DeForge 1980).

Birth Season: Lambing generally peaks in March in desert populations (Rubin et al. 2000) with most lambs born between January and March (NMDFG 2003), and May (occasionally April or June) in the remainder of range (Krausman et al. 1999, Shackleton et al. 1999).

Life history: Young are weaned in 4-6 months. Females first breed in their second year in southern populations; they may breed in their third year in north populations and occasionally breed in their first year in some areas (Krausman et al. 1999, Shackleton et al. 1999). The ability for females to reproduce begins to decline slightly after eight years of age (Caughley 1977). One of the most important habitat requirements for Bighorn is open, mountainous habitat in close proximity to escape terrain. Escape terrain is particularly important for Ewes during parturition and while raising young (NMDFG 2003). Ewes spend more time than bachelor rams in steep rocky escape terrain to maximize protection for themselves and lambs even though these areas generally have less abundant forage (NMDFG 2003)

D-1.3 Habitat and Ecology

D-1.3.1 Habitat

Terrain: Bighorn sheep habitat is typically made up of rough, rocky topography in mountains, foothills or river canyons (Shackleton et al. 1999, Krausman et al. 1999). Suitable escape terrain - steep slopes (i.e. cliffs, talus slopes) with a minimum of 60% slope that provide for predator avoidance is a crucial habitat requirement of desert bighorn sheep (NMDFG 2003). High quality and diversity of available forage plants are important to desert bighorn sheep, but it is noted that the amount of habitat available to sheep is limited by the amount of escape terrain close to open landscapes (McCarty and Bailey 1994). Terrain that allows for seasonal elevational movements between areas of foraging and breeding are also important (NatureServe 2013; USFWS 2008). Escape terrain is described as rock outcroppings and slopes of 27 – 85 % (Shannon et al. 2008). Rocky Mountain Bighorn in the north have been documented to spend as much as 86% of their time within 100 meters of escape terrain in winter (Oldemayer et al. 1971, Erickson 1972), and usually stay within 800 meters of escape terrain throughout the year (Pallister 1974). Desert bighorn use the micro-climates of cliff overhangs, granite outcroppings and slope aspect

to rest from direct sun in hot, dry environments (NatureServe 2013; San Diego Zoo Global Library 2009); solar heat on south aspects also reduces cold stress on bighorn sheep in northern habitats (Shackleton et al. 1999).

Elevation: Elevation varies considerably, both geographically and seasonally, from as low as 450 meters to over 3,300 meters (Shackleton et al. 1999).

Distance from accessible water: Northern populations of bighorn sheep do not usually require free-standing water, instead getting water from succulent vegetation in the summer and snow or ice in the winter (Van Dyke 1978). However, in the Madrean Archipelago region, bighorn have been shown to use water year-round (Gunn 2000) and in New Mexico, most bighorn are found within 1 mile of water (NMDFG 2003). Particularly during summer drought, bighorn may require accessible surface water to augment what moisture they are able to get from forage (Turner 1979, Turner and Weaver 1980, Seegmiller and Ohmart 1981). In the desert, bighorn can survive 10 or more days without drinking, and may meet some water needs by eating barrel cacti (Warrick and Krausman 1989; NatureServe 2013). Bighorn have been observed using free water in all New Mexico populations (NMDFG 2003). The presence of dependable water sources is strongly correlated with population persistence (Epps et al. 2004) and the creation of artificial water sources has been a component of sheep management and recovery for many years.

Vegetation: Bighorn sheep occur in mesic to xeric, alpine to desert grasslands or shrub-steppe (Shackleton et al. 1999, Krausman et al. 1999) and avoid low visibility areas with dense vegetation (Hall 1981, Hansen 1980, Risenhoover and Bailey 1985, Shackleton 1997, Ostermann et al 2001, USFWS 2000, San Diego Zoo). In Chihuahuan Desert habitat shrubs dominate their diet but their diet shifts with seasonal availability of newly emergent grasses and forbs during the summer-fall rainy season (Sandoval 1997, NMDFG 2003).

D-1.3.2 Mobility & Landscape Connectivity

In the Madrean Ecoregion, long-term persistence of desert bighorn may depend on movement corridors between populations (Bleich et al. 1990, Bailey 1992). They also rely on intermountain corridors for movement and to allow colonization of new areas or areas from which they were previously extirpated. Rams may also move long distances between mountain ranges in search of ewes during the rut (Geist 1971). In western Arizona, January-June home range of adult females was 19-27 square km (Seegmiller and Ohmart 1981). Male annual home range can be up to 37 square km in Nevada (Leslie and Douglas 1979).

D-1.3.3 Food

Desert bighorn sheep are opportunistic herbivores and ruminants. Turner (1973) documented 43 species of plants in diet. They are primarily grazers of grass and forbs, but their diet can also include significant amounts of shrubs and occasionally cacti, including *Acacia spp.*, *Encelia spp.*, sweetbush (*Bebbia juncea*), jojoba (*Simmondsia chinensis*) and *Krameria spp.* (Miller and Gaud 1989, Krausman et al. 1999, Shackleton et al. 1999). Diet can vary seasonally.

D-1.3.4 Phenology

Diurnal behavior. Feeding activity peaks at early morning and dusk, and grazing during the day is punctuated with periods of rest and rumination. Daily activity can vary by season; in winter, feeding periods are longer (NatureServe 2013).

D-1.3.5 Predators

Natural predators of the bighorn include coyotes (*Canis latrans*), bobcats (*Lynx rufus*), mountain lions (*puma concolor*), golden eagles (*Aquila chrysaetos*), gray wolves (*Canis lupus*). Coyotes have been shown to primarily prey young bighorn within a herd. In one case up to 80% of the year's lambs were lost to coyote predation (Hebert and Harrison 1988, Harper 1984, Hass 1989). Mountain lions, which are adapted to hunt in rugged terrain, are important predators in the Madrean Archipelago Ecoregion with mountain lions being documented as the cause of mortality in 85% of known-cause mortality in New Mexico desert bighorn (Harrison and Hebert 1988, Krausman et al. 1999, NMDFG 2003). Mountain lions have been cited as the primary factor preventing populations from recovering in New Mexico, not limitations of habitat quality or forage availability (NMDFG 2003).

Bighorn are known to alarm and startle easily at sudden animal movements and other disturbances (San Diego Global Library 2009). It is possible that fire suppression resulting in unnaturally dense vegetation cover may increase predation success (Etchberger et al. 1989).

D-1.3.6 Competitors

Carrying capacity for bighorn can be reduced through grazing by other ungulates, particularly domestic stock including cattle, sheep and burros. Mule deer (*Odocoileus hemionus*) are their primarily natural competitors (Shackleton 1985, USFWS 2000, San Diego Zoo Global Library 2009). Feral honey bees may interfere with sheep attempting to drink at artificial water sources (guzzlers) in desert environments (Boyce et al. 2003).

D-1.3.7 Demographics

Herds are matrilineal and based on female associations; Gregarious, but for most of the year adult males live apart from females and young (Shackleton et al. 1999, Krausman et al. 1999). Larger males less than 10 years old generally dominate younger males during the breeding season; males older than 10 years decline rapidly in condition. Bighorn tend to be faithful to their natal home range. Young learn from older sheep their home range, escape terrain, water sources and lambing habitat. Rams tend to range, moving between female ewe groups, and rams with larger horns and body size dominate both males and females; linear dominance orders among ewes (Shackleton 1985, USFWS 2000, Valdez and Krausman 1999).

D-1.3.8 Disease and parasites

Respiratory disease is the most serious disease affecting bighorn sheep. Disease was an important factor contributing to extinction of bighorn in much of their range and continues to limit populations today (Buchener 1960). As an example, contact with stray domestic sheep is believed to have resulted in the death (through bacterial pneumonia) of entire reintroduced herd of 65 in Warner Mountains, California and devastated bighorn populations in the mid and late 1900's (California Department of Fish and Game 1990). Respiratory disease can have large impacts on population dynamics of desert bighorn including all-age mortality events, high rates of mortality in lambs and chronic low-level infection in adults leading to sporadic mortality. Once introduced, respiratory pathogens can spread through interconnected populations for years. This is of particular concern for fragmented or isolated populations that may be pushed to extinction by stochastic events unrelated to disease (IDFG 2010).

Chronic sinusitis, pneumonia, and psoroptic scapies are the most common problems in bighorn sheep, and pathogens including contagious ecthyma virus, bluetongue, Pasteurella, contagious echthyma, infection keratoconjunctivitis, mycoplasma ovipneumoniae, and parainfluenza-3 virus have been found in declining populations. Decreased fitness occasionally occurs due to improperly fitted radio collars that cause physical injury to osseous and dermal tissues (Allen 1980, USFWS2000, Valdez and Krausman 1999, Clark et al. 1985).

D-1.4 Change Agent (CA) Characterization

The main change agents identified are forms of altered dynamics: Livestock management, development, and climate change.

Initial large declines were primarily the result of overhunting, habitat loss and competition with, and diseases and parasites introduced by, domestic stock including cattle, sheep and burros (Cowan 1940, Buechner 1960, Sugden 1961, Stelfox 1971, Goodson 1982, Boyce et al. 1990, Valdez and Krausman 1999). Urban encroachment, human disturbance and habitat changes resulting from removal of fire due to consumption of fine fuels by livestock and from fire suppression were likely factors in the disappearance of bighorn in southern Arizona's Catalina Mountain range in the 1990s, although disease or increased predation may have played a part (Etchberger et al. 1989).

D-1.4.1 Livestock management

In Idaho, introduced bighorns avoided cattle and were more sensitive to cattle presence than were established populations, based on previous studies (Bissonette and Steinkamp 1996). Decline of desert bighorn populations has been attributed to destruction of habitat and competition with domestic livestock and other native and domestic species (Seegmiller and Ohmart 1981).

Indicators: Landscape condition, active grazing leases, pasture management.

Stresses: increased competition, change in vegetation structure, decreased plant diversity, decreased forage abundance, increased behavioral disruption.

Responses: decreased physical fitness, decreased population size, reduced/ altered home range size.

D-1.4.2 Parasites and disease

Desert Bighorn are more sensitive to diseases and parasites than other native big game species. Desert bighorn populations have been heavily impacted by diseases introduced by livestock as people settled in the Madrean Archipelago Ecoregion. (NMDFG 2003). Psoroptic scabies from domestic sheep have been implicated in the decline of desert bighorn populations in the west (NMDFG 2003). Domestic sheep spread pneumonia, usually fatal for desert bighorn, throughout central and southern New Mexico (NMDFG 2003). In the Madrean Archipelago Ecoregion, even penned flocks of domestic sheep are of concern. Domestic goats were implicated in an outbreak of pneumonia in the Hell's Canyon Rocky Mountain bighorn (Cassirer et al. 1996). In the Arizona portion of the Madrean Archipelago Ecoregion, there was an incident of bighorn in the Silverbell Mountains dying from ecthyma and keratoconjunctivitis following contact with domestic goats. Cattle have not been conclusively shown to transmit disease to bighorn, but cattle do carry disease that are a potential source of bighorn disease (Bailey 1994) including bluetongue virus that is transmitted between ruminant species by a gnat (Luedke et al. 1967). Outbreaks of bluetongue have been documented in bighorn populations in New Mexico. Bunch et al. (1999) provides a general account of diseases and parasites affecting bighorn sheep which include chronic wasting disease, bacterial chronic sinusitis, contagious ecthyma, and parasitic elaeophorosis.

Indicators: Domestic stock with disease or parasite infestation, decreased recruitment, increased predation.

Stresses: Decreased fitness.

Responses: Decreased population size, reduced recruitment, decreased genetic health, increased mortality rate.

D-1.4.3 Habitat loss and alteration

Loss and alteration of habitat, especially key winter forage sites, is a key threat (Valdez and Krausman 1999, Shackleton et al. 1999, Krausman et al. 1999). Habitat degradation can occur through overgrazing by domestic stock, competition with exotic ungulates (e.g., Aoudad or Barbary, *Ammotragus lervia*), excessive off-road vehicle use, roads and trails, urban development, mining, spread of rangeland weeds, and the loss of water sources (Simpson 1980, Valdez and Krausman 1999, Krausman et al. 1999). Fire suppression and resulting vegetation succession leading to dense cover with the encroachment of tall dense shrubland and forest have been a major cause of habitat loss in Colorado and British Columbia (Davidson 1991, Cannings et al. 1999, Wakelyn 1987). Dense cover resulting from fire suppression is also a factor in herd declines in the Santa Catalina Mountains of Arizona (Etchberger et al. 1989). Fragmentation of habitat reduces or eliminates genetic interchange among populations (Ramey 2000) and reduces the probability of recolonization following local extirpation; both these effects are especially concerning in small populations (fewer than 100 individuals) vulnerable to extirpation (Berger 1990).

Indicators: Landscape condition, landscape permeability, habitat loss, groundwater level/ hydrology condition, distance from water, water quality, air quality, light pollution levels, noise index, human-wildlife conflict, distance from roads, wildlife-vehicle collisions, and direct take (hunting).

Stresses: Reduced habitat, decreased water availability, increased habitat fragmentation, loss of view shed/ increase in tall dense vegetation, increased behavioral disruption.

Responses: decreased population size, reduced recruitment, decreased home range, reduced migratory distance, decreased gene flow/ decreased genetic health, increased mortality rate.

D-1.4.4 Climate change

Bighorn rely greatly on specific habitat requirements to survive and thrive, which are affected by precipitation, temperature, seasonality, and other climatic conditions. Additional studies of these influences in consideration of a changing climate - including increasing temperature extremes, increased fire intensity and frequency, and drought - are needed, but some research outside the Madrean Archipelago ecoregion may provide insight to the responses that may be expected. Although bighorn are able to obtain much of the water that they need from their diet, the absence or reduction of succulent forage requires bighorn to seek out additional surface water. Decreased water resources and increased presence of predators at available water sites can expose bighorn to greater risk of mortality; in addition, warmer temperatures and lower precipitation may lead to increased water draw-downs by human populations (Heinz Center 2011, SWCCN 2008). The National Drought Mitigation Center (2013) reports that the Madrean Archipelago ecoregion has been experiencing Severe to Extreme drought in the last year, and forecasts this condition will persist or intensify.

Indicators: Increased drought, decreased water table/ groundwater level, decreased surface water, increased temperature extremes and duration, increased wildfire.

Stresses: increased change in vegetation structure, change in biotic community, decreased forage availability, decreased water availability, increased distance to water, increased predation, increased fire frequency and intensity, hyperthermia.

Responses: decreased population size, reduced recruitment, increased daily foraging movements, increased migratory distance, increased mortality rate.

D-1.5 Ecological Status: Key Ecological Attributes and Indicators

This section of the conceptual model addresses Key Ecological Attributes and their potential indicators. The ecological status is a way of describing current status via criteria, functionality, or levels of attributes and asks if they are within the normal range of variation.

D-1.5.1 Key Ecological Attributes

Table D-2 identifies the key ecological attributes for this CE within the Madrean Archipelago ecoregion. A **key ecological attribute** of a focal ecological resource is a characteristic of the resource's biology, ecology, or physical environment that is critical to the resource's persistence in the face of both natural and human-caused disturbance. Alteration of such a characteristic beyond some critical range of variation will lead to the degradation or loss of the resource within decades or less. The KEAs table lists the identified key ecological attributes, with a brief definition, a rationale for why it is important for the CE, and a listing of stressors or change agents that might be affecting the key attribute.

Table D-2. Key Ecological Attributes (KEA) used to determine the ecological status of desert bighorn sheep (*Ovis canadensis*) occurring in the Madrean Archipelago ecoregion.

KEA Class: Name	Definition	Rationale	Stressors
Landscape Context: Functional Landscape Connectivity & Landscape Permeability	The degree to which the landscape facilitates or impedes bighorn movement within and among habitat patches including connectivity of foraging habitat to escape terrain; and the relative potential for bighorn movement between populations at a regional scale. This may be indicated by the Landscape permeability index and Landscape Condition Model Index and/or assessing potential loss of key habitat association (escape terrain and foraging terrain).	At a local scale, permeability is critical for bighorn sheep to avoid predators and access food and water; at a larger landscape scale, it is critical for breeding between herds, dispersal, and migration between seasonal ranges. In the Madrean Ecoregion, long-term persistence of desert bighorn may depend on movement corridors between populations (Bleich et al. 1990, Bailey 1992). They also rely on intermountain corridors for movement and to allow colonization of new areas or areas from which they were previously extirpated. Rams may also move long distances between mountain ranges in search of ewes during the rut (Geist 1971). In western Arizona, January-June home range of adult females was 19-27 square km (Seegmiller and Ohmart 1981). Male annual home range up to 37 square km in Nevada (Leslie and Douglas 1979).	Stressors include fragmentation and habitat loss, caused by human development in the form of fencing, mining, urban development, and distance to roads, railroads and highways, among others. Fragmentation of habitat reduces or eliminates genetic interchange among populations (Ramey 2000) and reduces the probability of recolonization following local extirpation (Berger 1990). Low or absent landscape permeability can indicate locations of corridor bottlenecks and genetic isolation between populations that can lead to decreased genetic health, inbreeding and local extinction. Because bighorn have a strong flight and stress response to animal movement or disturbance (San Diego Global Library 2009), hiking trails and other areas frequented by humans or domestic dogs may also affect habitat permeability for bighorn sheep.
Size/Extent: Population Condition	Population size and demographics of bighorn herds, including sex and age ratios. This may also include evidence of population health (presence of disease or signs of chronic physical stress) which over time affect population size and demographics.	Population size and demographic structure directly indicates the viability and health of the population. Demographics can affect breeding systems, resulting in changes to genetic diversity; evidence of disease or signs of chronic stress may be a precursor to a decrease in population size. Initial large declines in bighorn sheep populations were primarily the result of overhunting, habitat loss and competition with, and diseases and parasites introduced by, domestic stock including cattle, sheep and burros (Cowan 1940, Buechner 1960, Sugden 1961, Stelfox 1971, Goodson 1982, Boyce et al. 1990, Valdez and Krausman 1999).	Stressors include causes of direct mortality such as predation, hunting, poaching, and disease; decreased fitness as the result of poor forage conditions, livestock use of forage, low water availability or flight/stress response to disturbance; and habitat fragmentation resulting in small habitat patches unable to support large herd populations. Urban encroachment, human disturbance and habitat changes resulting from fire suppression were likely factors to the disappearance of bighorn in southern Arizona's Catalina Mountain range in the 1990's, although disease or increased predation may have played a role (Etchberger et al. 1989).

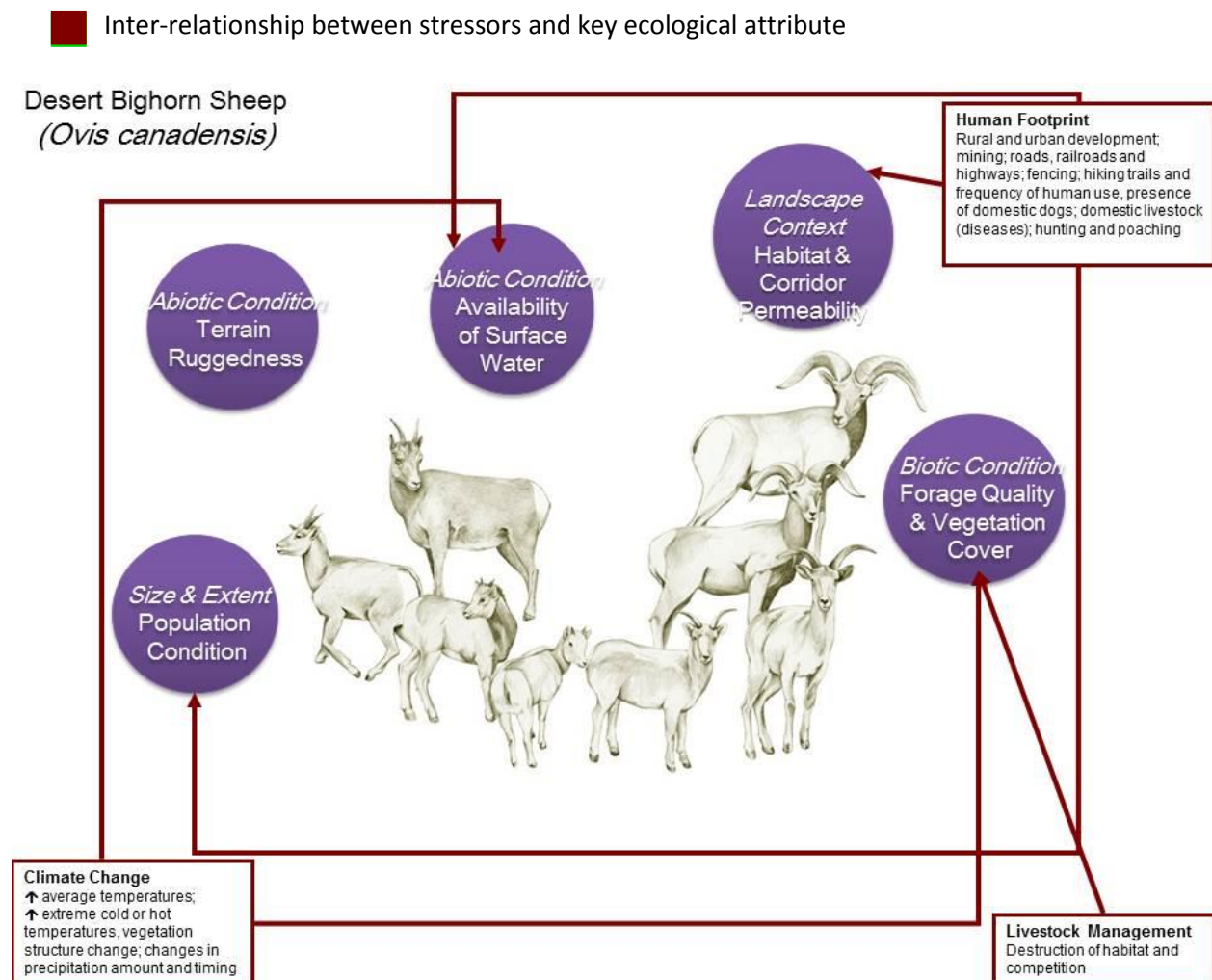
KEA Class: Name	Definition	Rationale	Stressors
Biotic Condition: Forage Quality & Vegetation Cover	Relative composition, abundance and species richness of native forage plants, average vegetation density and height within bighorn habitat areas.	Bighorn avoid low visibility areas with dense vegetation (Hall 1981, Hansen 1980, Risenhoover and Bailey 1985, Shackleton 1997, Ostermann et al 2001, USFWS 2000, San Diego Zoo). They are primarily grazers of grass and forbs, but their diet can also include significant amounts of shrubs and occasionally cacti, including <i>Acacia spp.</i> , <i>Encelia spp.</i> , sweetbush (<i>Bebbia juncea</i>), jojoba (<i>Simmondsia chinensis</i>) and <i>Krameria spp.</i> (Miller and Gaud 1989, Krausman et al. 1999, Shackleton et al. 1999). It is possible that unnaturally dense vegetation cover may increase predation success (Etchberger et al. 1989). In the Madrean Archipelago region, particularly during summer drought, or when surface water is less available, bighorn need to augment what moisture they are able to get from forage (Turner 1979, Turner and Weaver 1980, Seegmiller and Ohmart 1981). With high quality forage, bighorn are able to avoid frequent risk of predation at surface water sites.	Seasonal and monsoon precipitation has a large effect on forage quality; drought, invasive plants, livestock impacts to forage condition and fire suppression are also significant stressors on forage condition and vegetation density.
Abiotic Condition: Availability of Surface Water	Annual and seasonal precipitation and the resulting density, distribution and distance to surface water accessible by bighorn.	Madrean Archipelago region, bighorn have been shown to use water year-round (Gunn 2000) and in New Mexico, most bighorn are found within 1 mile of water (NMDFG 2003). Particularly during summer drought, bighorn may require accessible surface water to augment what moisture they are able to get from forage (Turner 1979, Turner and Weaver 1980, Seegmiller and Ohmart 1981). In the desert, bighorn can survive 10 or more days without drinking, and may meet some water needs by eating barrel cacti (Warrick and Krausman 1989; NatureServe 2013). Precipitation affects surface water availability and growth of high-nutrient forage.	Surface water sources include active seeps, springs, water tanks, wildlife waters, riparian and perennial streams, lakes and other sources of water in relation to bighorn habitat areas. Stressors affecting these open water sources include watershed development that alters runoff, infiltration (recharge), and evapotranspiration rates; surface water diversions, transfers, and use; groundwater withdrawals from basin-fill and alluvial aquifers; dams and dam operations; and riparian corridor development. Climate change also has the potential to cause increased drought and changes in human water use. Decreased water resources and increased presence of predators at available water sites can expose bighorn to greater risk of mortality.

KEA Class: Name	Definition	Rationale	Stressors
Abiotic Condition: Terrain Ruggedness	The topographic roughness (ruggedness) of the landscape or habitat area. A roughness index may be the standard deviation of slope, standard deviation of elevation, slope convexity, or degree of topographic contour.	Bighorn sheep habitat is typically made up of rough, rocky topography in mountains, foothills or river canyons (Shackleton et al. 1999, Krausman et al. 1999). Suitable escape terrain - steep slopes (i.e. cliffs, talus slopes) with a minimum of 60% slope that provide for predator avoidance is a crucial habitat requirement of desert bighorn sheep (NMDFG 2003). High quality and diversity of available forage plants are important to desert bighorn sheep, but it is noted that the amount of habitat available to sheep is limited by the amount of escape terrain close to open landscapes (McCarty and Bailey 1994). Terrain that allows for seasonal elevational movements between areas of foraging and breeding are also important (NatureServe 2013; USFWS 2008).	The availability of rugged terrain to bighorn is directly affected by human activity, including habitat loss through human encroachment and development. Climate change that alters high quality forage or water availability occurring in areas with ideal terrain may be a concern.

D-1.6 Conceptual Model Diagram

A conceptual model diagram for each CE provides a visual summary representation of the Key Ecological Attributes and species' life cycle (Figure D-2). These diagrams are intended to show how various stressors interact with categories of Key Ecological Attributes to highlight important drivers for species ecological integrity. The arrows indicate relationships between stressors and but do not indicate the nature of the influence (i.e. positive or negative).

Figure D-2. Conceptual model for the desert bighorn sheep, showing key ecological attributes (by class) for this species, and indicating relationships between stressors and KEAs. Original artwork by Jane Kim.



D-1.7 References for the CE

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D-2 Black-tailed Prairie Dog (Cynomys ludovicianus)

D-2.1 Classification Comments

Prairie dogs are rodents within the squirrel family (*Sciuridae*); they occur only in North America. There are five species of prairie dogs in North America: the black-tailed prairie dog (*Cynomys ludovicianus*), the white-tailed prairie dog (*C. leucurus*), the Gunnison's prairie dog (*C. gunnisoni*), the Utah prairie dog (*C. parvidens*), and the Mexican prairie dog (*C. mexicanus*) (Pizzimenti 1975). The Utah and Mexican prairie dogs are currently listed as threatened (49 FR 22339) and endangered (35 FR 8495), respectively. Generally, the black-tailed prairie dog occurs east and north of the other four species in more mesic habitat.

The black-tailed prairie dog was first described by Ord in 1815 from a specimen local to the Upper Missouri River (Hall and Kelson 1959); they were extirpated by 1961, although the species was first collected by members of the Lewis and Clark expedition of 1804-1806. The species is found in eastern Montana, eastern Wyoming, eastern Colorado, eastern New Mexico, southwestern North Dakota, western and central South Dakota, western and central Nebraska, western and central Kansas, western and central Oklahoma, northwestern Texas, and in a small area of south-central Canada, and was reintroduced to Arizona in 2008.

Figure D-3. Black-tailed prairie dog (*Cynomys ludovicianus*). Photo courtesy Arizona Game and Fish Department website, www.azgfd.gov.



D-2.2 Biology and Distribution Status

Distribution – North America: The historic range of the black-tailed prairie dog included the Great Plains from southern Saskatchewan south to the Texas panhandle, and southern New Mexico and Arizona into extreme northern Mexico (Prairie dog coalition, 2013). Today it occurs from extreme south-central Canada to northeastern Mexico and from approximately the 98th meridian west to the Rocky

Mountains. Black-tailed prairie dogs are absent from most of the range which they occupied historically, although remnant populations are widely scattered across much of this area. Notably, many land tracts within the historic range have no black-tailed prairie dogs, some have a few black-tailed prairie dogs, and very few have significant numbers of black-tailed prairie dogs (USFWS 2000).

The extirpation of most historic populations (and burrow systems), vegetative community changes (e.g., brush invasion of grasslands), landscape changes (e.g., cropland conversion, urbanization), and the establishment of sylvatic plague in North America may limit any extensive reoccupation of its former range by the species. Additionally, most historic black-tailed prairie dog occupied habitat no longer exists and source populations for recolonization are often isolated (USFWS 2000).

Madrean Archipelago ecoregion:

Black-tailed prairie dog occupied habitat existed in southeastern Arizona and New Mexico prior to rodent control efforts (Hall and Kelson 1959). Black-tailed prairie dogs occur on the Coronado National Forest of Arizona (Patton, 1994).

Information provided by Van Pelt (1999) noted the species in Cochise and Graham Counties. Information from Fagerstone and Ramey (1996) as delineated by the Black-footed Ferret Recovery Foundation noted the species in Cochise and Santa Cruz Counties as well as portions of Pima, Graham, Pinal, and Greenlee Counties. As of August 2013, the results of a reintroduction program launched by the Arizona Game and Fish Department in 2008, there are four small colonies within the Las Cienegas National Conservation Area (Treadwell, personal communication, 2013).

By 2011, three sites within Las Cienegas NCA had been stocked, but due to extreme draught, each of these colonies had experienced population declines from high predation and low forage availability. To maintain stability and genetic integrity, eighty animals from two sources (New Mexico and Sonora, Mexico) were released at these three sites. It was also determined by a University of Arizona survivorship study conducted in 2011, that the draught conditions were forcing the prairie dogs to travel well outside their colonies and into the tall grass to search for food. This was resulting in high predation, causing the small populations to decline rapidly. It was also determined that annual offspring numbers were low. From 2009-2011, a total of 34 pups emerged, with only 10 in 2011. To combat these issues, the project decided to provide supplemental feed. Initially, the additional food succeeded in reducing predation mortalities. In 2012, the feeding was begun earlier in March with the hope of increasing the production of offspring. The result was a virtual population explosion when 132 pups emerged. This technique is now standard operating procedure for new colony establishment. This extra production also allowed the program to source animals for the fourth colony in 2012 from the first three (Treadwell, personal communication, 2013).

In New Mexico, black-tailed prairie dogs occupied the desert grasslands in most of the state. Specific reports by county vary as to the occurrence of the species. The range of the subspecies *C.l. arizonensis* overlaps with that of *C.l. ludovicianus* in Torrance, Chaves, and Lincoln counties (Schmitt 1995).

In 1996, surveys on black-tailed prairie dogs were conducted in New Mexico, including Colfax, Union, Harding, and Mora Counties, covering about 11,500 square miles (Van Pelt 1999). These surveys revealed the presence of a total of 1,191 black-tailed prairie dog individuals in over 41 colonies in Union County, 11 colonies in Colfax County, two colonies in Mora County, and 10 colonies in Harding County. Similar surveys were conducted in 1997 by Paternoster, revealing the presence of 33 colonies of black-tailed prairie dogs in Roosevelt County and 40 colonies in Curry County together totaling about 3,174

occupied acres. Although these two surveys revealed a total of 137 colonies of black-tailed prairie dogs in six counties, current systematic surveys such as these are lacking in the remaining counties of New Mexico (Van Pelt 1999).

Given some gaps in current information concerning their distribution, numbers, viability of isolated populations, impacts of habitat fragmentation, amount and extent of efforts to control (or eradicate) them, population impacts from plague, and other biological factors, the overall status of black-tailed prairie dogs in New Mexico needs further verification. However, there are no data available that would suggest populations in New Mexico are stable and/or improving (Van Pelt 1999).

D-2.2.1 Population

The U.S. Fish and Wildlife Service notes the unavailability of any recent, comprehensive, original, single-source, range wide estimate of occupied habitat for the black-tailed prairie dog and recognizes that such an evaluation would be very difficult to conduct (USFWS, 2000). Several authors have attempted to combine reports of occupied habitat from various sources to provide an overall estimate across the species' range (Fagerstone and Ramey 1996, Knowles 1995, Knowles 1998, Mulhern and Knowles 1995). However, these efforts have been limited because the range of the species is vast and much of it is remote, and accordingly accurate, comparable information is difficult to obtain. Populations are located irregularly, at varying densities, and may be periodically expanding and/or contracting over time and space due to various combinations of factors.

Most estimates of prairie dog population trends are not based on numbers of individual animals, but on estimates of the amount of occupied habitat. Estimates of black-tailed prairie dog density vary seasonally, but range from 2 to 18 individuals per acre (5 to 45 per hectare) (Fagerstone and Ramey 1996, Hoogland 1995, King 1955, Koford 1958, Miller et al. 1996). Most prairie dog surveys do not estimate density because of the associated effort and cost. The US Fish and Wildlife Service believes that a review of various estimates of black-tailed prairie dog occupied habitat provides the best available and most reasonable means of determining population trends and the status of the species (USFWS, 2000).

The black-tailed prairie dog may be found intermittently in remnant populations throughout much of the range that it once occupied. However, Barko (1997), Fagerstone and Ramey (1996), Knowles (1998), Mulhern and Knowles (1995), and Wuerthner (1997) concluded that there has been an approximate 94-99 percent reduction in the amount of black-tailed prairie dog occupied habitat within this range since about 1900 (USFWS, 2000). Black-tailed prairie dogs had the most extensive range of all the species of prairie dogs; they probably occupied more area than all other species combined (Hoogland 1995).

At present, the black-tailed prairie dog may be found scattered in remnant populations throughout much of the range that it once occupied. A significant portion of existing black-tailed prairie dog occupied habitat rangewide occurs in a few large complexes. Approximately 36 percent of the remaining occupied habitat for the species in North America occurs in seven complexes, each larger than 10,000 acres (4,000 hectares). In 2000, the USFWS believed that approximately 768,000 acres (311,000 hectares) of black-tailed prairie dog occupied habitat existed range wide (USFWS, 2000).

The Arizona Game and Fish Department first released 74 black-tailed prairie dogs in 2008 from the Ladder Ranch in New Mexico at the Las Cienegas National Conservation Area. At least four offspring were identified the following spring. An additional 107 prairie dogs were released in 2009 at the previous release site and a new release site which is also in the Las Cienegas National Conservation Area.

Due to the sensitive nature of the species during re-establishment, the hunting season for black-tailed prairie dogs is closed. Risks to the species need to be minimized while the population grows and expands in its historic range (AZGFD 2013).

In New Mexico, where *Cynomys ludovicianus* was identified as a species of greatest conservation need in the Comprehensive Wildlife Conservation Strategy for New Mexico (NMDGF, 2006). Formerly they were widespread and abundant east of the Rio Grande and in the grasslands of southwestern New Mexico. Colonies were often reported in marginal habitat, such as open woodland, and in the southwestern part of the state they occupied semidesert conditions (Findley et al. 1975).

In Northern Mexico, the population size of black tailed prairie dogs is unknown, however estimates conducted in 1990 in the Pancho Villa - Monteverde - Loma de los Ratones y Buenos Aires, Chihuahua, was between 258,000 and 380,000 individuals. In 1993, it was estimated there were over one million individuals distributed in a complex of populations dispersed in approximately 55,000 hectare (136,000 acres); being at that time the largest population in North America (CONABIO, 2011). According to the Arizona Game and Fish Department, there are still two extant colonies in Sonora, Mexico, just south of Las Cienegas NCA in Arizona. In 2011, AGFD and CEDES personnel completed density mapping at these colonies using the Blgens et al method. The La Mesa colonies had 177.2 acres, and the population estimate wa 1,351 individuals with a 95% confidence interval of 931 to 1,771 animals. The Las Palmitas colony had 146 acres, with a population estimate of 1,905 individuals and 1,440 to 2,371 animals at the 95% C.I. (Treadwell, personal communication, 2013).

D-2.2.2 Reproduction

Breeding Systems: Black-tailed prairie dogs are not prolific in comparison to many other rodents. Several biological factors determine the reproductive potential of the species. Females usually do not breed until their second year and live 3-4 years (Hoogland 1995, King 1955, Knowles and Knowles 1994). Survival of young prairie dogs can be high in some circumstances, especially in low density populations where habitat resources are plentiful and repressive factors such as control or disease are not operative (Garrett et al. 1982); although much lower rates of annual increase or even reductions in colony size can occur where vegetation hinders expansion or constricts existing colonies (Osborn and Allan 1949), or due to high predation rates.

Mating is polygynous, with the usually single male mating with multiple females within his coterie. In cases when there is more than one resident male, usually brothers, females will mate with both. Reproduction occurs once per year in spring, although the timing varies with latitude, and females are typically sexually receptive for only one day of the year (Hoogland 1995) though if conception fails they can undergo a second estrous.

Young are born blind, naked and mostly helpless (Hoogland 1995), and do not emerge from the burrow until around six weeks of age, and are weaned shortly after that (Davis and Schmidly 1994). Interestingly, after emerging from the burrow, but prior to the end of lactation, pups may nurse from females other than their own mother, an example of 'cooperative breeding' (Hoogland 1995; Texas Parks and Wildlife 2006). Females remain in their natal coterie for life, while males disperse before their first breeding season (Hoogland 1995). Likewise, adult males rarely remain within the same coterie for more than two breeding seasons, probably to reduce the possibility that they will mate with their own female offspring. Females can live up to eight years of age, whereas males tend not to live longer than five years in the wild (Hoogland 1995).

Gestation: 33-38 days. Pups are born in April or May.

Litter size: Female black-tailed prairie dogs produce a single litter, usually 4-5 pups, annually (Hoogland 1995, Knowles and Knowles 1994). Other authors mention an average litter size of 3 to 4 pups with the number of pups ranging between 1 and 8 (USFWS, 2000). One female may produce from 0 to 20 young in its lifetime.

Breeding Season: February through March.

Birth Season: April or May.

D-2.2.3 Life history

Prairie dogs are small, stout ground squirrels. The total length of an adult black-tailed prairie dog is approximately 14-17 inches; the weight of an individual ranges from 1 to 3 pounds, males being larger than females. Individual appearances within the species vary in mixed colors of brown, black, gray, and white. Their black-tipped tail and long claws in the front legs for digging are characteristic (CONABIO 2011, Hoogland 1995, USFWS 2000).

Black-tailed prairie dogs are diurnal, burrowing animals. Individuals spend most of the day above ground. They do not hibernate as do white-tailed, Gunnison's, and Utah prairie dogs but may remain underground during bad weather, with a month or more being documented during a severe winter period (Hoogland 1995, Tileston and Lechleitner 1966). The species is very social, living in population aggregations called colonies, towns, or villages (Hoogland 1995, IUCN 2013). Historically, they generally occurred in large colonies that contained thousands of individuals, covered hundreds of thousands of acres, and extended for miles (Bailey 1905). Most existing colonies are much smaller. When unsuitable habitat such as a hill, tall vegetation, or a stream divides a prairie dog colony, the resulting sub-colonies are called wards (King 1955). Within colonies, prairie dogs live in territorial, harem-polygamous family groups called coteries (Hoogland 1995).

Black-tailed prairie dogs exhibit a high degree of social organization, living in enormous colonies known as 'towns' containing from hundreds to millions of individuals (Hoogland 1995, IUCN 2013). Each colony shares an elaborate network of burrows for shelter and protection against predators, often covering areas of 100 hectares or more (IUCN 2013), with the largest ever recorded colony covering 65,000 square kilometers and containing an estimated 400 million animals (Hoogland 1995, IUCN 2013). Colonies are subdivided into 'wards', and then into smaller family units called 'coteries', populated by a group of closely related females, one or two territorial males, and any offspring under two years of age (Hoogland 1995, Texas Parks and Wildlife 2006). Members of a coterie share food supplies outside of the breeding season and cooperate to aggressively defend their territory from neighbors (Hoogland 1995, Smithsonian 2006). However, while males respond strongly to intrusion by other males they seem oblivious to invading females; females, by contrast, show the most hostility toward invading females. During the breeding season, females aggressively defend their natal burrow against other females and, given the opportunity, will even raid the burrows of other females and kill their pups (Hoogland 1995).

The colonial nature of prairie dogs, especially the black-tailed prairie dog, is a significant characteristic of the species. Hoogland (1995) described the sociality, demography, and population dynamics of the black-tailed prairie dog. Coloniality offers an effective defense mechanism by aiding in the detection of predators and by deterring predators through mobbing behavior. It increases reproductive success through cooperative rearing of juveniles and it aids parasite removal via shared grooming. However, it

has been noted that coloniality promotes the transmission of disease, which can significantly suppress populations (Olsen 1981, Hoogland 1995). Accordingly, disease may play a major factor in the population dynamics of the species.

D-2.3 Habitat and Ecology

Many authors have recognized the biological importance of the black-tailed prairie dog as a keystone species (Agnew et al. 1986, Ceballos and Pacheco 1997, Clark et al. 1982, Kotliar et al. 1999, Miller et al. 1994, Reading et al. 1989). Keystone species influence ecosystem functions through their activities in unique and significant ways. The ecological effect caused by a keystone species is disproportionate to its numerical abundance and its removal or decline initiates changes in ecosystem structure and a decline in overall species diversity (Kotliar et al. 1999, Mills et al. 1993, Paine 1980, Power et al. 1996, Terborgh 1988). However, Stapp (1998) questioned whether the black-tailed prairie dog is truly a keystone species. He recognized various ecological values of the species, but challenged other authors' view of the overall role of the species. Kotliar et al. (1999) concluded that prairie dogs provide some unique functions compared to other herbivores in the system and that continued decline of the species may lead to a substantial erosion of biological diversity; and, therefore, keystone status is appropriate. The extent to which these interrelationships directly affect the black-tailed prairie dog itself is largely unknown.

Prairie dogs act in several roles inasmuch as they are prey, provide shelter, modify vegetation, and influence ecological processes in a manner not entirely duplicated by other prairie herbivores (Ceballos and Pacheco 1997, Kotliar et al. 1999, List et al. 1997, Miller et al. 1994, Wuerthner 1997). While the black-tailed prairie dog creates habitat for itself and other species, it also is affected by other species. For example, prairie dogs can create preferential grazing opportunities for herbivores that in turn create opportunities via grazing for the expansion of prairie dog colonies at their perimeters. However, the degree to which the black-tailed prairie dog itself is influenced by these and other prairie species, particularly ungulates, is not well understood. For example, the removal of large numbers of bison (*Bison bison*) and other native ungulates from the North American prairie may have had effects on the ecology of the black-tailed prairie dog that can no longer be fully evaluated. Similarly, the periodic effects of fire no longer influence much of the remaining fragmented prairie environment.

D-2.3.1 Habitat

The black-tailed prairie dog lives in arid and semi arid deserts and grasslands of North America (CONABIO, 2011). Family groups live in burrows in dry prairies with short grass (<30 cms is preferred). Their burrows have an entrance that is surrounded by a pile of dirt, looking like a 'volcano' protecting it from floods and providing good visibility to defend against predators.

Recent studies demonstrate that areas in the North American Great Plains with prairie dog colonies support higher densities of at least 9 vertebrate species than sites without colonies. Prairie dogs affect habitat for these species through multiple pathways, including creation of belowground refugia, supply of prey for specialized predators, modification of vegetation structure within colonies, and increased landscape heterogeneity (Augustine and Baker 2013).

Terrain: Prairie dogs prefer flat lands, with little to no slope, well-drained soils and few rocks (CONABIO, 2011). Roe and Roe (2003) describe the ideal soil as <30% clay, ~50% sand and >70% silt. In Mexico, Black tailed prairie dogs are found between 1400 and 1600 meters of elevation (Pacheco and Ceballos 2005). Rangelwide, they occur between 700 and 2200 meters elevation.

Distance from accessible water: The prairie dog gets most of the water it needs from the plants it eats.

Vegetation: Black tailed prairie dogs forages selectively from the plants available in its habitat. Diet also varies seasonally. In the summer black tailed prairie dogs prefer to feed upon wheatgrass (*g. Agropyron*), buffalo grass (*g. Bromus*), grama (*g. Bouteloua*), rabbitbush (*g. Chrusothamnus*), and globemallow (*g. Sphaeralcea*). In the winter they eat prickly pear cactus (*g. Opuntia*), thistles (*g. Cirsium*), and various roots (Hoogland 1995).

D-2.3.2 Landscape connectivity

Prairie dog dispersal is usually limited to approximately 3 miles (5 kilometers) or less, and individuals dispersing from home colonies generally move into an established colony rather than attempting to initiate a new colony (Garrett and Franklin 1988, Hoogland 1995). Black-tailed prairie dogs avoid certain landforms and soils, and preferentially select some landforms and some soils. These limitations could restrict recruitment of animals into small and declining isolated populations and favor reestablishment of individuals in nearby, recently abandoned colonies over the establishment of new colonies. Prairie dogs use human-made land alterations, like roads, to assist in landscape-level emigration and discovery of potential new habitat for colonization. King (1955) observed two types of emigration among black-tailed prairie dogs. The first type occurred when yearling males move in the spring following the appearance of young in the colony. The second type of emigration consisted of older adults leaving a colony in the spring or summer, possibly to avoid excessive attention and interference by the young-of-the-year.

D-2.3.3 Food

Black-tailed prairie dogs mainly consume grasses, sedges, forbs (flowering plants), roots and seeds, though they are also known to eat grasshoppers, beetles and other insects (USFWS 2000). In northern Mexico, prairie dogs consume grasses such as *Agropyron* spp., *Bromus* spp., and *Bouteloua* during summer and in winter they consume cactus *Opuntia* spp., herbs (*Cirsium* spp.), shrubs (*Chrysanthamnus* spp.) and roots (CONABIO 2011). The animals usually eat all the plants right around its burrow.

D-2.3.4 Phenology

Black-tailed prairie dogs are diurnal, burrowing animals. Sunrise to 0900 and 1500 to sunset are the peaks of activity on a colony during mid summer (Clark et al. 1971 in Keinath 2004). This pattern is largely driven by the temperature at ground level during these months (June-August), which can be too hot for extended activity above ground (Grant 1995 in Keinath 2004). However, in early season (February-April) and late season (September – November) the peak of activity is unimodal and occurs around 1300 (Clark et al. 1971 in Keinath 2004). Activity is most common between the temperatures of 15° to 70° F, although short bouts of activity can occur between the temperatures of 75° to 80° F. Activity bouts can be affected by high wind speeds (exceeding 11 m/s) (Grant 1995 in Keinath 2004).

Daily behaviors of white-tail prairie dog observed by Orabona-Cerovski (1991) indicate that spend about one third of their time above ground feeding ($x = 36\%$) and one third sitting erect ($x = 33\%$) possibly scanning for predators. Sitting horizontally accounts for 16% of the daily activity. The remainder of the behaviors (running, vocalizing, fighting, playing, kissing, digging, grooming) made up the remaining 15% of time spent above ground (Orabona-Cerovski 1991 in Keinath 2004).

Mating season runs from February through March. A month after mating, the female will have three to four pups. The pups are born naked and with their eyes closed. They stay in the burrow for about six weeks. They are weaned when they are about seven weeks old but will stay near their mother for another two weeks. Colony activity is at its highest around May and will not begin to decline until late

July, when adult males begin to disappear below ground. Adult females will then begin to descend into burrows two to three weeks later (Clark 1973).

In the fall, prairie dogs put on a layer of fat to help them survive in the winter. Black tails do not hibernate as do white-tailed, Gunnison, and Utah prairie dogs (Hoogland 1995, Tileston and Lechleitner 1966), but when the winter weather is extremely cold or snowy, they may go into a light hibernation-like sleep and stay in their burrows for a few days.

D-2.3.5 Predators

Although reports vary as to those species that require prairie dogs for their survival, at least 9 species depend directly on prairie dogs or their activities to some extent, and another 137 species are associated opportunistically (Kotliar et al. 1999). The most obligatory species of this group is the black-footed ferret (*Mustela nigripes*). Probably no other species has a more clearly documented dependence on another species than does the black-footed ferret on the prairie dog (Anderson et al. 1986, Biggins et al. 1986, Clark 1989, Forrest et al. 1988, Henderson et al. 1974, Hillman 1968, Miller et al. 1996). The black-footed ferret is a federally listed endangered species that depends upon prairie dogs as a source of food and uses its burrows for shelter. Any actions that kill prairie dogs or alter their habitat could prove detrimental to black-footed ferrets occupying the affected prairie dog towns.

Other predators include the coyote (*Canis latrans*), kit fox (*Vulpes macrotis*), swift fox (*Vulpes velox*), gray fox (*Urocyon cinereoargenteus*), badger (*Taxidea taxus*), bobcat (*Lynx rufus*), mountain lion (*Puma concolor*), long-tailed weasel (*Mustela frenata*), ferruginous hawk (*Buteo regalis*), burrowing owl (*Athene cunicularia*), Golden eagle (*Aquila chrysaetos*), Bald eagle (*Haliaeetus leucocephalus*), and reptiles such as the Gopher snake (*Pituophis melanoleucus*) and rattlesnakes (*Crotalus* spp.) (CONABIO 2011; USFWS 2000). Clearing away the plants around the burrow helps the prairie dog spot predators.

D-2.3.6 Disease and parasites

Sylvatic plague—an exotic disease that entered North America in 1900—threatens the survival of prairie dogs. The influence of sylvatic plague on black-tailed prairie dog populations has had significant depressant effects on remnant populations since the late 1900s (USFWS 2000). However, given that about 10% of the historic range is both plague-free and available, limited immune response has been observed in some individuals, and some sites have demonstrated the ability to recover to pre-plague levels, the USFWS has concluded that plague no longer appears to be as significant a threat as previously thought, and is not likely to cause the black-tailed prairie dog to become an endangered species in the foreseeable future (USFWS 2004). In Arizona, sylvatic plague may not occur at elevations below 4,500 feet (1,372 meters), where most of the species occurred historically (Van Pelt, 1999, USFWS, 2000). Bubonic plague from Europe transmits very easily to colonies and wipes them out fast (Prairie Dog Coalition 2013).

Common Black tailed prairie dog parasites include fleas (*Opisocrostis hirsutus*, *O. tuberculatus*, *O. labis*, *Pulex simulans*, *P. irritans* and *Leptopsylla segnis*), mites (*Ixodes kingi* and *Atricholaelaps glasgowi*) and lice (CONABIO 2000, IUCN 2013).

D-2.3.7 Threats

Four major impacts have had a substantial influence on black-tailed prairie dog populations. The first major impact on the species was the initial conversion of prairie grasslands to cropland across its whole range from approximately the 1880's through the 1920's. The second major impact on the species was large-scale control efforts conducted from approximately 1918 to approximately 1972 in efforts to reduce competition between prairie dogs and domestic livestock and to minimize destruction of

agricultural crops by prairie dogs. The third major impact on the species was the inadvertent introduction of an exotic disease, sylvatic plague, from the Old World into North American ecosystems around 1900, with the first recorded impacts on the black-tailed prairie dog in 1946 (USFWS 2000). The fourth major impact on the species is habitat fragmentation, which is a serious threat at the local level due to inbreeding. Additionally the remaining colony could be heavily impacted by natural or human-related catastrophic events such as plague or illegal shooting.

Humans pose the greatest threat to prairie dogs, frequently poisoning and shooting the animals and often plowing or bulldozing entire colonies for cropland or development. Many ranchers dislike the animals because they eat grass that ranchers would rather have for their livestock (CONABIO 2011). As a result, the former range and numbers of the black-tailed prairie dog have been dramatically reduced, and the considerable reduction in population numbers has also seriously threatened, amongst others, the black-footed ferret, for which they were virtually sole prey.

The main reason black-tailed prairie dogs are no longer found in Arizona is due to extensive extermination efforts that were initiated in the early 1900s. Prairie dogs were considered pests and ranchers viewed them as competition with their cattle for grass. While massive poisoning efforts are no longer occurring in Arizona, extermination still occurs in many places within its' range. Poisoning not only kills prairie dogs but, depending on the type of poison, often kills some animals that eat the poisoned prairie dog. Continued use of poisons could not only reduce prairie dog numbers, but could also cause a decline in the populations of the animals that prey upon them (AZGFD 2013).

Conservation status – The Black tailed prairie dog is classified as 'Least Concern' (LC) on the IUCN Red List; 'Rare' in the United States and 'Threatened' in Mexico.

The Arizona Game and Fish Department (AZGFD) classifies all prairie dogs native to the State, black-tailed and Gunnison's, as nongame mammals. In 1999, the hunting season for black-tailed prairie dogs was closed (Van Pelt 1999). Arizona does not require the eradication of prairie dogs for agricultural purposes or promote recreational shooting of prairie dogs (Van Pelt 1999). The black-tailed prairie dog is listed as Species of Greater Conservation Need. This list is being modified into a State list of "Wildlife of Special Concern" and AZGFD has proposed the black-tailed prairie dog for inclusion on the new list (Van Pelt, Arizona Game and Fish Department, pers. comm. 1998; USFWS 2000).

In New Mexico the black-tailed prairie dog was identified as a species of greatest conservation need in the Comprehensive Wildlife Conservation Strategy for New Mexico (NMDGF, 2006).

D-2.4 Change Agent Characterization

This section of the conceptual model presents a narrative description of the primary change agents and current knowledge of their effects on Black tailed prairie dog populations. The main change agents identified are the destruction and modification of habitat, disease and pet trade and shooting.

D-2.4.1 Destruction and modification of the black tailed prairie dog's habitat

Black tailed prairie dog habitat destruction and modification to croplands is the biggest and most widespread change agent for the species across its distribution range. Significant destruction, modification, and curtailment of black-tailed prairie dog habitat and range have occurred for many years. By the end of the 1990s, approximately 33 percent of the historic range of the black-tailed prairie dog and 37 percent of the suitable habitat within its range had been converted to cropland uses in the United States. This fundamental land use change resulted in significant destruction of black-tailed prairie

dog habitat, mostly in the eastern portions of the species' range where adequate precipitation favored farming. The present threat of large-scale destruction of black-tailed prairie dog habitat through cropland conversion is much less than in the early days of agricultural development due to the fact that land with the highest potential for traditional farming uses was converted many years ago. However, conversion of rangelands and prairie habitat to cropland still occurs in some areas due to continuing improvements in intensive agricultural techniques (USFWS 2000).

Although efforts to control prairie dogs continue in New Mexico, the New Mexico Department of Game and Fish (NMDGF) believes these control activities are now more localized in application and on a smaller scale than in the recent past. Control of prairie dogs is still done for the purpose of protecting grazing lands, agricultural crops, and farming developments. Other control activities, usually in close proximity to human developments, include eradication or control of prairie dogs for their presumed role in transmission of sylvatic plague to humans. Management of black-tailed prairie dogs has not been limited to control. Black-tailed prairie dogs have recently been reintroduced in two areas in Sierra County, and two areas in Hidalgo County are being discussed for possible reintroduction in areas of private ownership on the Gray Ranch (Van Pelt 1999).

In Mexico, List et al. (1997) reported that occupied black-tailed prairie dog habitat in Mexico declined by 34 percent between 1988 and 1996, in part due to rangeland conversion due to farming. In Arizona, the AZGFD noted a 35 percent reduction in grassland habitat along the San Pedro River to mesquite woodland invasion that could be due to the cumulative effects of fire suppression, grazing practices and perhaps the elimination of the black-tailed prairie dog (USFWS 2000).

Within the MAR, black-tailed prairie dog habitat and regional historical ranges extend north and south of the U.S.-Mexico border. Prairie dog populations occurring in the Madrean ecoregion can be affected by physical barriers to gene flow, recruitment, behavior, and habitat loss and destruction, caused by border infrastructure and other border-related activities.

D-2.4.2 Disease

The Fish and Wildlife Service believes that sylvatic plague is likely the most important factor in recent reductions of many black-tailed prairie dog populations throughout a significant portion of the range of the species. Approximately 66 percent of the species' range has been affected by plague (USFWS 2000). Sylvatic plague is a disease caused by the bacterium *Yersinia pestis*, which fleas acquire from biting infected rodents and other species and then transmit via a bite. The disease also can be transmitted directly between animals. The term "sylvatic" refers to the occurrence of the disease in the wild. It also may be referred to in its bubonic, pneumonic, or septicemic forms, depending on the affected portion of the organism in which it is observed. Given the communicability and lethality of plague, an epizootic may affect an entire colony in a similar manner as a pathogen may affect an individual animal. An entire black-tailed prairie dog colony may disappear just as an individual black-tailed prairie dog would die from a plague infection. Plague, once established in an area, becomes persistent and periodically erupts, with the potential to extirpate local black-tailed prairie dog populations (USFWS 2000).

D-2.4.3 Pet trade and shooting

Herron (Texas Parks and Wildlife Department, pers. comm. 1999; USFWS 2000) and others have reported that black-tailed prairie dogs are removed from the wild for sale as pets. Herron was aware of three commercial operators who collectively removed approximately 5,000 individuals from the Texas panhandle and other States annually in the 1990s, although these efforts may have declined. One animal export company in Texas noted that over the past 4 years their company has bought and sold

approximately 20,000 black-tailed prairie dogs, largely from the same locations in western and northwestern Texas (Shaw et al. 1993). Miscellaneous reports indicate that this practice occurs elsewhere in the species' range, but the extent of removal of individuals from the wild for use as pets is unknown (USFWS 2000).

One factor impacting black-tailed prairie dog populations in some local areas is recreational (sport or varmint) shooting. Extensive shooting, especially of pregnant females or females nursing young, could significantly reduce annual recruitment and change the ultimate population dynamics of a colony. Gross estimates of the number of modern shooters of prairie dogs and their potential take, based on reports from the field, suggest that hundreds of thousands of black-tailed prairie dogs are probably shot across their range annually. Small local populations already depressed by disease and other adverse influences may suffer shooting impacts as additive losses. Shooting impacts also may contribute to population fragmentation and preclude or delay recovery of colonies reduced by other factors such as sylvatic plague.

D-2.5 Ecological Status: Key Ecological Attributes and Indicators

This section of the conceptual model addresses Key Ecological Attributes and their potential indicators. The ecological status is a way of describing current status via criteria, functionality, or levels of attributes and asks if they are within the normal range of variation.

D-2.5.1 Key Ecological Attributes

Table D-3 identifies the key ecological attributes for this CE within the Madrean Archipelago ecoregion. A **key ecological attribute** of a focal ecological resource is a characteristic of the resource's biology, ecology, or physical environment that is critical to the resource's persistence in the face of both natural and human-caused disturbance. Alteration of such a characteristic beyond some critical range of variation will lead to the degradation or loss of the resource within decades or less. The KEAs table lists the identified key ecological attributes, with a brief definition, a rationale for why it is important for the CE, and a listing of stressors or change agents that might be affecting the key attribute.

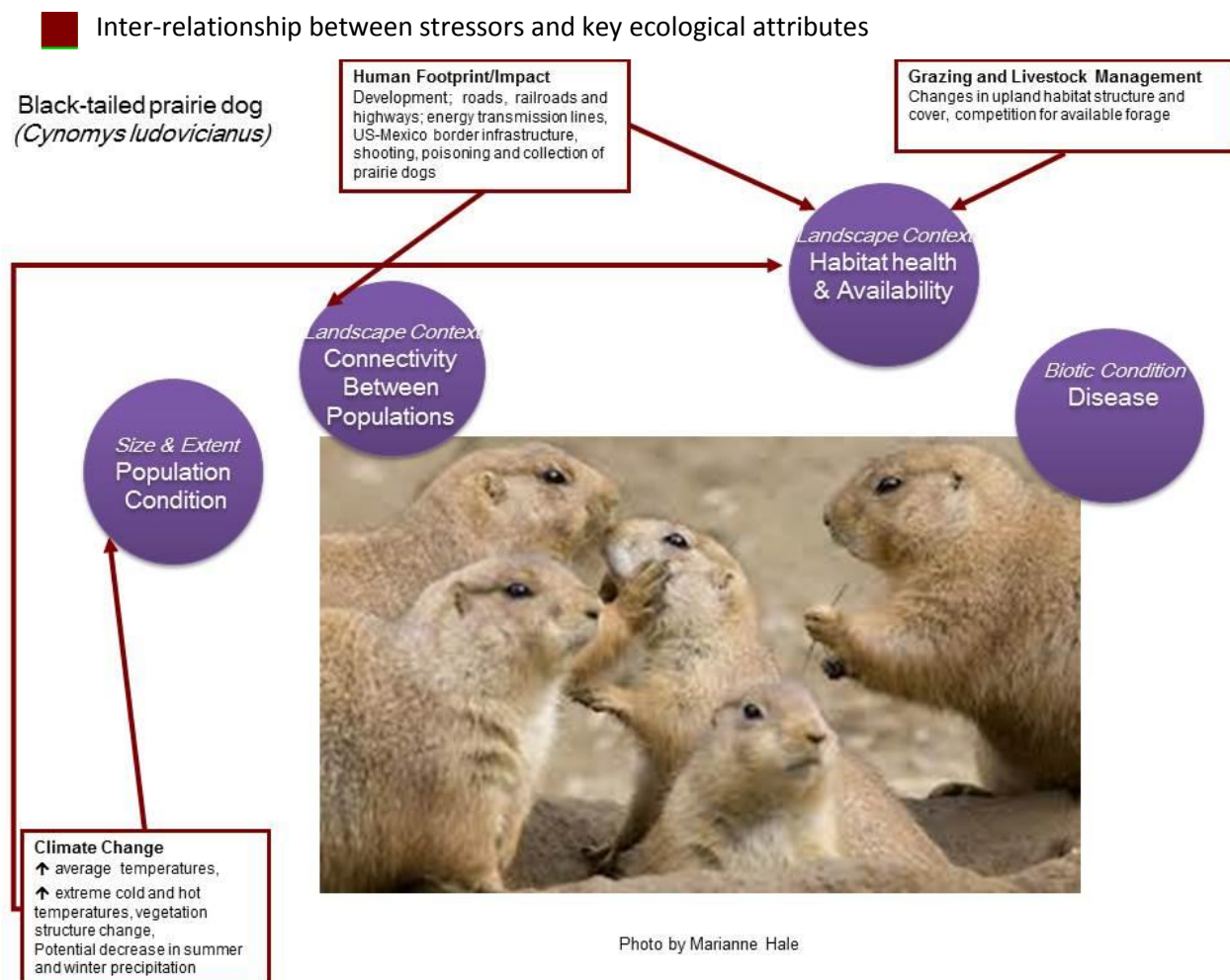
Table D-3. Key Ecological Attributes (KEA) used to determine the ecological status of Black-tailed Prairie Dog (*Cynomys ludovicianus*) occurring in the Madrean Archipelago ecoregion.

KEA Class: Name	Definition	Rationale	Stressors
Size/Extent: Population Condition	The size and age and sex distribution of prairie dog populations.	Prairie dog population size and demographic structure indicates the viability and health of a population, metapopulation or prairie dog colony. Females do not breed until their second year and reproduction occurs once per year during spring.	Shooting of females, pregnant females or females nursing young can significantly reduce annual recruitment. Extermination efforts by humans had previously extirpated prairie dogs from Arizona.
Landscape Context: Habitat health and availability	The extent of intact arid and semi arid deserts and short-grass grasslands suitable for prairie dogs.	Significant destruction, modification, and curtailment of black-tailed prairie dog habitat and range have occurred for many years. Black tailed prairie dog habitat destruction and modification to croplands is the biggest and most widespread change agent for the species across its distribution range. Prairie dogs prefer flat lands, with little to no slope, well-drained soils and few rocks (CONABIO, 2011). Roe and Roe (2003) describe the ideal soil as <30% clay, ~50% sand and >70% silt. In Mexico, Black tailed prairie dogs are found between 1400 and 1600 meters of elevation (Pacheco and Ceballos 2005).	Historically, an important stressor for established prairie dog populations has been diminishing or destroying their habitat, mainly due to conversion to agriculture and cattle grazing (USFWS 2000). Livestock management acts as a change agent for prairie dogs through habitat destruction, change of vegetative communities, and direct mortality due to eradication control efforts. Land use change results in significant destruction of black-tailed prairie dog habitat.
Landscape Context: Connectivity between populations	The connectivity between prairie dog populations along US-Mexico borderlands	Black-tailed prairie dog habitat and regional historical ranges extend north and south of the U.S.-Mexico border. Prairie dog populations occurring in the Madrean ecoregion can be affected by physical barriers to gene flow, recruitment, behavior, and habitat loss and destruction, caused by border infrastructure and other border-related activities.	US-Mexico Border infrastructure & border related activity physical reduce or prevent movement between prairie dog populations north and south of the border that were historically connected.
Biotic Condition: Prairie dog health and disease	Disease	The Fish and Wildlife Service believes that sylvatic plague is likely the most important factor in recent reductions of many black-tailed prairie dog populations throughout a significant portion of the range of the species. Approximately 66 percent of the species' range has been affected by plague (USFWS 2000).	Sylvatic plague is caused by the bacterium <i>Yersinia pestis</i> , acquired and transmitted by fleas.

D-2.6 Conceptual Model Diagram

A conceptual model diagram for the CE provides a visual summary representation of the Key Ecological Attributes and species' life cycle (Figure D-4). These diagrams are intended to show how various stressors interact with categories of Key Ecological Attributes to highlight important drivers for species ecological integrity. The arrows indicate relationships between stressors and but do not indicate the nature of the influence (i.e. positive or negative).

Figure D-4. Conceptual model for the black-tailed prairie dog, showing key ecological attributes (by class) for this species, showing key ecological attributes (by class) for this species, and indicating relationships between stressors and KEAs.



D-2.7 References for the CE

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D-3 Coues White-tail Deer (*Odocoileus virginianus couesi*)

D-3.1 Taxonomic or Classification Comments

Endemic to North America, Coues white-tail deer (*Odocoileus virginianus couesi*) or Venado Cola Blanca in Mexico, one of 38 subspecies of White-tailed deer found in the new world, is found primarily in central and southeastern Arizona, southwestern New Mexico and north central Mexico. The white-tailed deer was first described in Virginia by Thomas Hariot in 1584 (Heffelfinger 2006). The Coues White-tail deer was first classified as (*Odocoileus virginianus couesi*) in 1874 by U.S. Army surgeon Dr. Joseph Rothrock, who collected two specimens in 1874 and suggested they be called *Odocoileus virginianus couesi* for the naturalist Dr. Elliot Coues. Dr. Coues never actually collected specimens (Carmony 1985). The Coues white-tailed deer is most closely related to *O. v. Texanus*, with which it may exhibit some interchange with populations in Durango, Mexico (Findley 1975). There is some controversy regarding the designation of subspecies due to the fact that classification is currently based primarily on morphological characteristics (e.g. size, pelage color, size and shape of male antlers) (Hoffmeister 1986).

White-tailed deer range from 60 degrees north latitude in Central Canada to 15 degrees south latitude in South America: and from the east coast of the United State and Canada to the West Coast. In the United States and Canada there are 16 sub species. White-tailed deer have not been described in California, Nevada, and Utah (Hesselton and Hesselton 1982). There are 14 sub species found in Mexico with only Baja and parts of Sonora and Chihuahua void of the species. With 38 identified subspecies, 14 found in Mexico and 16 in Canada and the United States, and a north/south range between 60 degrees north in Canada and 15 degrees south in Peru; the white-tailed deer is the most widely distributed and studied cervid in the American Continent where it inhabits wide variety of habitat types throughout its range (Mandujano 2010, Geist 1998).

For the Madrean Ecoregional assessment we shall focus exclusively on one subspecies, the Coues white-tail deer (*Odocoileus virginianus couesi*).

References to the *Texanus* and other subspecies are occasionally used to understand unique adaptation responses relevant to this arid borderlands ecoregions. Within this ecoregion populations of Coues white-tailed deer in Arizona are currently managed and monitored in at least 17 different Game Management Units (AZGFD 2013). In the New Mexico portion of the Madrean Archipelago Ecoregion the Coues white-tail deer occurs in areas of New Mexico Game Management Units (GMUs) 15, 16, 17, 21, 22, 23, 24, 26, and 27; aerial surveys have been flown in some of the GMUs each year to develop trend data (NMGF 2013). Populations of Coues white-tailed deer in Northern Sonora are managed through established seasons and bag limits. There is no evidence that they are regularly monitored or surveyed.

Figure D-5. Coues white-tail deer (*Odocoileus virginianus couesi*).



D-3.2 Biology and Distribution Status

D-3.2.1 Distribution

Mixed populations of mule deer (*Odocoileus Hermionus*) and Coues white-tailed deer (*Odocoileus virginianus couesi*) occur within the Madrean Archipelago Ecoregion today. The Coues White-tailed deer is distributed through out the Madrean Archipelago including both north and south of the border (Hoffmeister 1986). Coues white-tailed deer occur from near the Mogollon Rim in Central Arizona, to south of the Mexican/U.S. border in Durango, Mexico where they tend to blend with other subspecies (Galindo-Leal 1993, Knipe 1977). Isolated populations are found as far east as the Black Range, Gila, Animas, Alamo Hueco, San Luis, Magdalena and Peloncillo Mountains of southwestern New Mexico and as far west as the Ajo Mountains in Organ Pipe Cactus National Monument in western Arizona (Lopez 2006). A single subspecies, Coues white-tailed deer, occurs in Arizona (Hoffmeister 1986). While the species is not usually thought of as a desert animal, the Coues white-tailed deer inhabits virtually every Mountain range in Sonora east of the 113th meridian, including the arid sierras de Alamo, Jojoba, Picu', and los Mochos (Carmony and Brown 1991). A prominent food source, the deer is highly regarded as source of protein and features prominently in the Vigida (Vikita) ceremony of the Hia C'ed and Tohono O'odham Their east/west range extends from the foothills of the Sierra Madre in Western Durango, to the far west with a population in the Sierra Cubabi 8km south of Sonoita on the NE side of Mexico Highway 2. (Felger & Broyles, *Dry Borders: Great Natural Reserves of the Sonoran Desert*).

D-3.2.2 Home Range

Home range is vital to Coues white-tailed deer. It must be large enough to provide essential components of food, shelter, and water, but small enough to be totally familiar to deer for survival advantages (Ockenfels 1991). There have been limited studies designed to define the home range of Coues white-tailed deer. The best estimate of Coues white-tailed deer home range comes from research done by Ockenfels et. al. (1993). The overall area they used during that time and more intensive "core area" estimated does had a home range of averaging 2 mi² (range = .02 – 7.0 mi²) with a core area of more intensive use of 0.7 mi². As is typical, bucks had larger home ranges, averaging 4.1 mi² (range=1.7-6.5 mi²) with a core area of 1.7 mi² (Heffelfinger 2006). Coues white-tailed deer do not need to migrate, but do have seasonal movements related to cover, food and water availability (Ockenfels, 1991).

D-3.2.3 Population

Arizona's statewide Coues white-tailed deer population is based on the sum of population estimates gathered from hunt management units; these are produced with computer simulation models using data gathered from aerial surveys and annual hunter-reported harvest data. Fixed-wing aircraft, helicopter and ground surveys are conducted, prior to hunting season. The current population of Coues white-tailed deer in Arizona is estimated at 82,000 post hunt adults (AZGFD 2013). New Mexico has a smaller population size in relation to Arizona, however, Coues populations have been expanding and increasing the past few years based on biologists observations. Mexico has abundant populations but there are limited surveys and no known estimate of the current size of the Mexican population.

D-3.2.4 Reproduction

Breeding Systems: Local fluctuation in reproductive success are related to seasonal and annual reproductive phenology; antler development, rutting behavior, and antler cast for males, while rutting season, fawn drop, and lactations are important time periods for females. Rutting season is partially related to photoperiod, the ratio of diminishing daylight to darkness (Verme and Ullery 1984). Rut takes place earlier in northern white-tailed deer populations beginning in September and continuing for several months terminating in January. Variation in rut timing is related to latitude with southern populations occurring up to 2 months later than northern populations. The rut for Coues White-tailed

deer is normally from mid-December into March (McCabe and Leopold 1959, Welles 1959). The Tohono O'odham honor this important source of food by indicating February as Uhwaliig Machath, "moon of the deer-mating odor" (The Antler cast for most populations starts in late December and peaks in January. Antler cast generally occurs after breeding season and is based on decreasing levels of testosterone in males. Female white-tailed deer are receptive to breeding only 2-3 days during each estrus cycle (Knox et al. 1988) For Coues white-tailed deer, peak of estrus should coincide with the rut (Ockenfels 1994).

Gestation: The gestation period for Coues white-tailed deer is between 200 and 207 days with fawning from mid-July through mid-September, peaking in late July or August (Knipe 1977). However timing can vary from year to year based on timing of conception and nutrition of the doe. Very poor nutrition can delay breeding or lengthen the gestation period (Verme 1965).

Litter size: One to two fawns, per breeding female per year. Twins are the norm after the first year with single fawns the norm in the first breeding cycle.

Breeding Season: Coues white-tailed deer breed between the months of November and March with the peak rut taking place in mid-January. Many of the extremes are likely young, inexperienced buck. The average date of conception is early to mid January (Ockenfels 1991)

Birth Season: Depending on habitat and environmental conditions, Coues white-tailed deer generally give birth in the Madrean Archipelago ecoregion between mid-July and mid-September. Fawning usually peaks between late July and August. (Heffelfinger 2006)

Life history: Coues white-tailed deer weigh 4 to 6 pounds at birth, which is approximately 10 to 15% of the adult female's weight. Young are weaned by 2 to 3 months but already feed on vegetation during the first month. Rapidly growing fawns lose their spots after about two months. Males reach sexual maturity in 1.5 years but may not have the opportunity to breed until they are dominant enough to compete with older males. Females are sexually mature at 18 months. Females often fail to conceive in their first breeding cycle. The life span of the Coues white-tailed deer is approximately 10 years but they can live to be 20 (Hoffmeister 1986, USFS 2013).

D-3.3 Habitat and Ecology

D-3.3.1 Habitat

Knowledge of Coues white-tailed deer habitat requirements are mostly observational in nature. Coues white-tailed deer have several specific habitat requirements needed for survival.

Food: Shrubs, Forbs, Grasses, Mast, Succulents

Cover: Escape cover, Bedding and Fawning cover, Thermal cover

Space: Activity center, Home range

Water: Surface water, Preformed water (Fulbright & Ortega-s 2006)

Elevation: In the Madrean Archipelago Coues white-tailed deer are found between elevations ranging from 3,000 to 10,000 feet, with the greatest densities of deer concentrated within the 3,500 to 5,500 feet elevation.

Terrain: Coues white-tailed deer occupy rolling hills, bajadas and deep canyons with adequate water sources, food and over story. They seek terrain that decreases the risk of predation and dehydration

under extreme conditions, deer select habitats based on the degree of cover. At lower elevations the *Prosopis* habitat provides the greatest amount of hiding and thermal cover and edible food. In order to detect and avoid or escape approaching predators Coues white-tailed deer tend to prefer habitats and bedding areas that allow for adequate visibility (Bello et al. 2001). Both bucks and does prefer slopes of 20 –29 degrees for resting and feeding, usually on the upper third of major and minor ridges. Habitats offering adequate hiding and thermal cover during extremes in summer temperatures are critical to Coues white-tailed deer survival (Ockenfels 1994).

Distance from accessible water: Coues white-tailed deer require regular access to surface water and regularly travel one and a half miles to water during the dry season in May and June (Knipe 1977). Coues white-tailed deer tend to move to permanent water during the dry season where they will stay until the rains return later in the year. The distance to water sources varies between years and between seasons. Coues white-tailed deer select habitat within one half mile of a permanent water source, but show a distinct affinity for areas within one quarter of a mile, with very few animals located more than three-quarters of a mile from water (Ockenfels 1991).

Vegetation: Coues white-tailed deer prefer mountainous habitat, associated with Madrean Evergreen Woodlands (Brown & Lowe 1982). They are found primarily in woodland communities consisting of evergreen oaks or in mixed oak-pinion-juniper. They are also found in Ponderosa pine forests, in desert scrub, in deciduous forests and occasionally in spruce-fir (Hoffmeister 1986)

D-3.3.2 Landscape connectivity

In the Madrean Archipelago region Coues white-tailed deer do not migrate in the traditional sense. But they do show seasonal shifts in habitat induced by the availability of food, water, cover, and the effects of weather hunting, breeding, livestock grazing and other factors. These seasonal migrations are limited within separate herd home ranges and are not as impressive as long-range migrations found in many other ungulates. However, these seasonal movements are essential to maintaining gene flow and finding quality forage and water sources in summer and winter (Hefflefinger 2006). Coues white-tailed deer will move higher or lower in their home range depending on temperature and snow cover. These deer are generally more abundant on southern and eastern exposures at 6000 to 7,000 feet in winter reflecting the seasonal movements associated with temperature and snow depth. Coues white-tailed deer are similar to mule deer in that the rut tends to drastically change activity patterns with large-scale movement taking them out of their home range. (Welch 1960).

D-3.3.3 Food

Coues white-tailed deer require a diversity of vegetation for forage and thrive with higher plant species richness. They are opportunistic herbivores; shrubs and browse makes up the highest proportion of their diet. Coues white-tailed deer change their forage preferences season to season and with habitat type and elevational gradient. They will seek the most nutritious and succulent forage species available. In the lower elevation western deserts they consume: jojoba, fairy duster, mallow and agave while in the central highlands of Arizona and at higher elevations they forage on: desert ceanothus, ratany sugar sumac, white sage and holly leafed buckthorn. Coues white-tailed deer populations found in southeastern Arizona are known to eat a wide range of shrubs and succulents including: velvet-pod mimosa, ocotillo, fairy-duster, barrel cactus, buckwheat, Findler's ceanothus, oaks, alligator juniper berries and leaves, Kidney wood, silk tassel and prickly poppy (Hefflefinger 2006). In Arizona forbs, shrubs, trees make up the majority of food with cacti, grasses and other plants accounting for minor dietary intake throughout the year (Ockenfels 1991).

D-3.3.4 Phenology

Coues white-tailed deer activity patterns are circadian, crepuscular and diurnal. Movement patterns show animals active between 8 am and 10 am and again between 4 pm and 6 pm during the winter months (Krauseman and Ables 1981). During the summer 70% of the deer observed by Knipe (1977) between 9 am and 5 pm were bedded. (Heffelfinger 2006). Activity patterns for Coues white-tailed deer are variable, and influenced by region, weather, season, and also by herd dynamics. High intensity hunting pressure has been shown to induce seasonal movements in Coues white-tailed deer (Welch 1960).

D-3.3.5 Predators

The native prey of the Mexican gray wolf was almost entirely the Coues white-tailed deer. The Arizona wolf would have fed on the white-tailed deer and mule deer. (Brown 1983) With limited populations of Mexican Wolves in Coues white-tailed deer habitat the two most important predators in the Southwest are Mountain Lions (*Felis concolor*) and Coyote (*Canis latrans*). Mountain lions are the most important predator on adult deer, where as coyote can kill a large number of young fawns (Ballard et al. 2001). Other predators include black bears (*Ursus americanus*), bobcats (*Lynx rufus*), golden eagles (*Aquila chrysaetos*), and possibly an occasional transient jaguar (*Panthera onca*) (Heffelfinger 2006).

D-3.3.6 Competitors

Coues white-tailed deer compete with other browsers and grazers on the landscape. Presence of cattle in whitetail habitat may induce temporary avoidance movement, however, competition with livestock is minimized by the fact that the higher elevations are not grazed as heavily as the lower elevations that would likely be used more intensively by mule deer. Welch (1959) reported that White-tailed deer actively avoid cattle and in some parts of the southwest overgrazing in dry years is still a problem for whitetails and their habitat (Galindo-Leal et al. 1997).

D-3.3.7 Demographics

Sparse or small populations can result in variations in sex ratios, age distributions, and birth and death rates, which can cause fluctuations in population size. Males tend to stay separate from females during most of the year only coming together to breed. In very small isolated populations, males may have trouble finding females leading to decreased recruitment. The composition of a herd can change hourly or day to day, forming loosely dispersed groups that keep in visual contact when food is readily available. During times of stress, access to adequate food is the main factor in lower recruitment and mortality (Heffelfinger 2006).

D-3.3.8 Disease and parasites

Hemorrhagic disease has been identified occasionally in free-ranging whitetail deer, mule deer and bighorn sheep in Arizona (Noon et al. 2002) Coues white-tailed deer are susceptible to any number of diseases. The most common are caused by infection with either bluetongue or epizootic hemorrhagic disease. Adenovirus caused hemorrhagic disease, papillomas or fibropapillomas, foot and mouth disease, vesicular stomatitis, bovine viral diarrhea, and other miscellaneous viruses infect whitetails. Bacterial infections can cause sickness in Coues white-tailed deer including; dermatitis, bovine tuberculosis, lumpy jaw, leptospirosis, brain abscess and bovine brucellosis. Chronic wasting disease can infect white-tailed deer and has been documented in south-central New Mexico in close proximity to Coues white-tailed deer populations but not in Arizona.

Parasites are common in deer populations but may become only significant when deer are under stress. Nasal bots, mites, ticks louse flies, flea flies and screwworm are known external parasites. Internal parasites include elaeophorosis, commonly called carotid worm; foot/worm is very common,

gastrointestinal nematodes of stomach worm, abdominal worm lungworm and tapeworm (Heffelfinger 2006)

D-3.4 Change Agent Characteristics

This section of the conceptual model presents a narrative description of the primary change agents and current knowledge of their effects on Coues white-tailed deer populations. The main change agents identified are forms of altered dynamics: Livestock management, development, and climate change.

D-3.4.1 Livestock grazing and management

Competition for food cattle is one stressor on Coues white-tailed deer especially during already poor forage conditions. Ockenfels (1997) found that under moderate grazing regimes and “normal precipitation, there is little evidence of forage competition between deer and cattle. However there is evidence that cattle stress deer during times of drought or in areas where overgrazing is prevalent. Heavy livestock use coupled with frequent drought periods reduces available browse (Knipe 1977) can reduce perennial grass cover needed for fawn survival. (D. Brown 1984, M. Brown 1984). Competition between deer and cattle can also occur for non-forage resources. Heffelfinger (2006) describes the dynamic of removal of forage in a large area surround livestock waters. The placement of new waters on the landscape may more negatively affect deer via forage removal than the positive effect from the new source of water. Coues white-tailed deer main food sources are shrubs, and to a lesser extent trees (Knipe 1977). Gallina et al. (1981) found that shrubs were the highest percentage (51%) of Coues' year-round diet, and together with trees accounted for 83% of deer diet in Mexico. Grasses constitute only a minor part of deer diet but could be important to deer in spring and year round in certain areas. Grazing use should be based on actual current grass production instead of a set AUM allotment or a range analysis that is not indexed to climatic conditions prior to measurement (Ockenfels 1994).

Stresses: increased competition during drought periods, change in vegetation structure, decreased plant diversity, decreased forage abundance, decreased water quality, increased soil compaction, habitat degradation, increased behavioral disruption.

Responses: decreased population size, reduced home range size, increased seasonal movement, reduced recruitment, increased mortality rate.

D-3.4.2 Development

Human development influences Coues white-tailed deer by directly affecting home range, habitat and resources, and may fall into the categories of 1) Military and Border-defense related activity; 2) Mining; 3) Rural and urban development; 4) Transportation in the form of roads, railroads and highways; and 5) Energy Development including new transmission lines, canals, and converting open space for solar and wind farming. These stressors are present in the Madrean Archipelago ecoregion where Coues white-tailed deer occur. Border infrastructure, including lighting, roadways, and frequent vehicle traffic, pose a significant barrier to wildlife movement north and south of the border and wherever they may transect home ranges. Large-scale mining can have a tremendous impact on ground water resources, water and air quality, and habitat availability. Rural and urban develop has a direct impact on ground water levels and habitat resources, and transportation and energy development results in habitat fragmentation and direct mortality from vehicle collisions. Increased human development also infers increased human presence: access to more remote areas for recreation and hunting may increase direct take from hunting, road kill, and other disturbance.

Stresses: reduced habitat/ open space, decreased water availability, increased habitat fragmentation, decreased water quality, habitat degradation, increased behavioral disruption, increased migratory barriers.

Responses: decreased population size, reduced recruitment, decreased home range, reduced migratory distance, decreased gene flow/ decreased genetic health, increased direct mortality rate, increased mortality rate of the population.

D-3.4.3 Climate change

Coues white-tailed deer rely greatly on specific habitat requirements to survive and thrive. These habitat requirements are affected by precipitation, temperature, seasonality, and other climatic conditions. Additional studies of these influences in consideration of a changing climate including increasing temperature extremes, increased fire intensity and frequency, and drought are needed. Although Coues white-tailed deer are able to obtain much of the water that they need from their diet, the absence or reduction of succulent forage requires deer to seek out additional surface water. Decreased water resources and increased presence of predators at available water sites may expose deer to greater risk of mortality, although O'Brien et al. (2006) found that contrary to common perception, rates of predation at human-made water catchments is relatively low so this may not be an issue. In addition, warmer temperatures and lower precipitation may lead to increased aquifer draw downs by human populations due to both decreased infiltration and increased demand for water (Heinz Center 2011, SWCCN 2008). The National Drought Mitigation Center (2013) reports that the Madrean Archipelago ecoregion has been experiencing "Severe to Extreme" drought in the last year, and forecasts this condition will persist or intensify.

Stresses: increased change in vegetation structure, change in biotic community, decreased forage availability, decreased water availability, increased distance to water, increased predation, increased fire frequency and intensity, greater potential for disease transmission, average temperatures, hyperthermia.

Responses: decreased population size, reduced recruitment, increased daily foraging movements, increased seasonal movement distance, increased mortality rate.

D-3.5 Ecological Status: Key Ecological Attributes and Indicators

This section of the conceptual model addresses Key Ecological Attributes and their potential indicators. The ecological status is a way of describing current status via criteria, functionality, or levels of attributes and asks if they are within the normal range of variation.

D-3.5.1 Key Ecological Attributes

Table D-4 identifies the key ecological attributes for this CE within the Madrean Archipelago ecoregion. A **key ecological attribute** of a focal ecological resource is a characteristic of the resource's biology, ecology, or physical environment that is critical to the resource's persistence in the face of both natural and human-caused disturbance. Alteration of such a characteristic beyond some critical range of variation will lead to the degradation or loss of the resource within decades or less. The KEAs table lists the identified key ecological attributes, with a brief definition, a rationale for why it is important for the CE, and a listing of stressors or change agents that might be affecting the key attribute.

Table D-4. Key Ecological Attributes (KEA) used to determine the ecological status of Coues white-tailed deer (*Odocoileus virginianus couesi*) occurring in the Madrean Archipelago ecoregion.

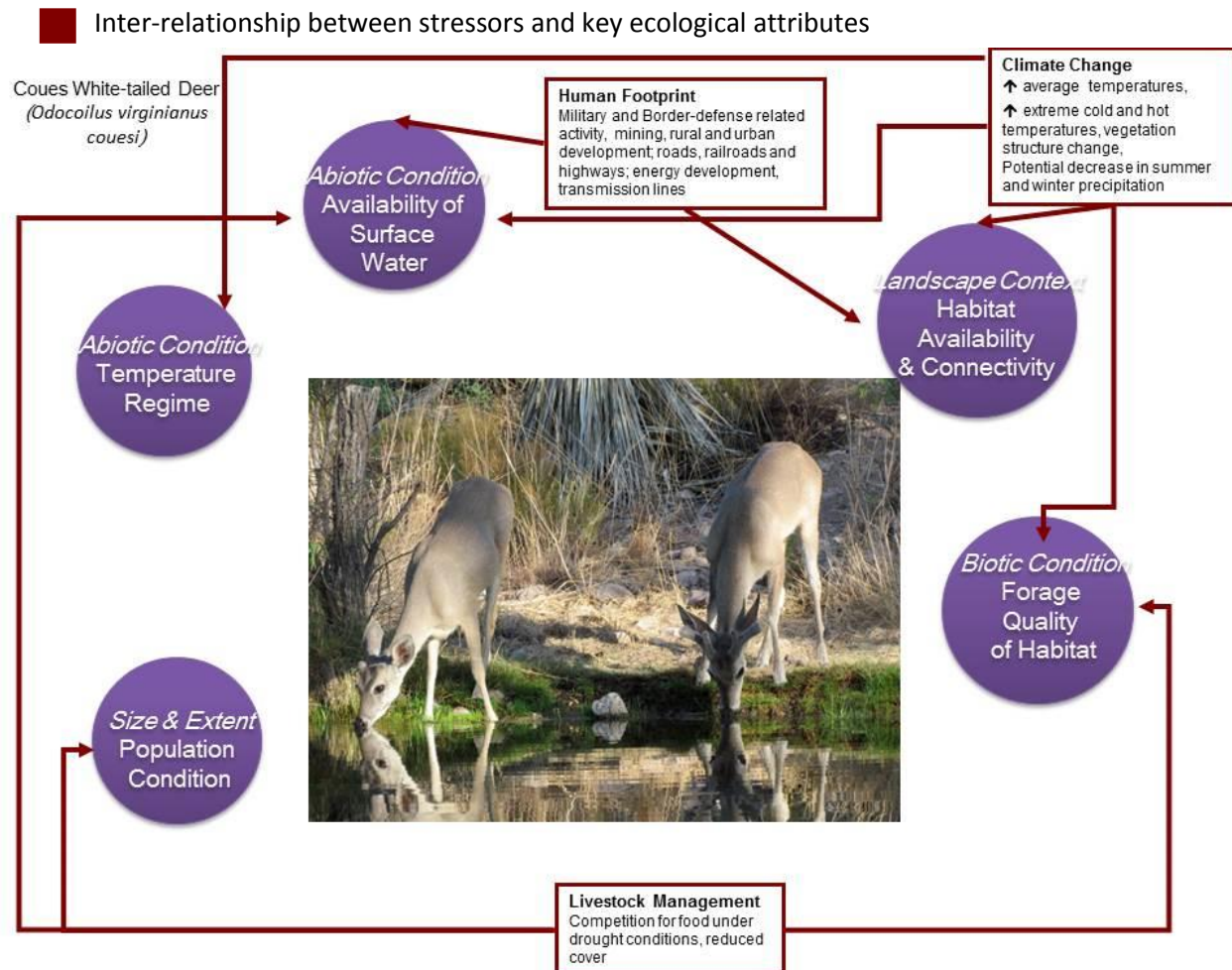
KEA Class: Name	Definition	Rationale	Stressors
Landscape Context: Habitat Availability and Connectivity	The extent and connectedness of mountainous Madrean evergreen woodland and mixed oak-pinyon juniper habitat that is intact.	Daily and seasonal movement between habitat patches within home range is essential for Coues white-tailed deer adaptability to changing forage and water availability, predator avoidance, and gene flow. Coues white-tailed deer prefer habitats with topographic ruggedness of between 20 and 29 degrees of slope (Ockenfels 1994). This habitat preference may restrict movement and home range size as water and forage conditions essential to Coues white-tailed deer survival shift or change on an elevational gradient. Coues white-tailed deer prefer mountainous habitat, associated with Madrean Evergreen Woodlands (Brown 1984). They are found primarily in woodland communities consisting of evergreen oaks or in mixed oak-pinyon-juniper. They are also found in Ponderosa pine forests, in desert scrub, in deciduous forests and occasionally in spruce-fir (Hoffmeister 1986).	Stressors include destruction and fragmentation of habitat by human development. Coues white-tailed deer habitat and historical herd home ranges extend north and south of the U.S.-Mexico border. White-tail populations occurring in the Madrean ecoregion are directly affected by the physical barriers to gene flow, behavior disturbance, and habitat loss and destruction, among other stressors, caused by border infrastructure and other border-related activities. Increased road density, direct loss of foraging habitat and increased human interaction as well as disturbance of diurnal bedding sites can have a negative impact on Coues white-tailed deer populations (Ockenfels 1994).
Size/Extent: Population Condition	Population size and demographics of Coues white-tailed deer herds, including sex and age ratios. This may also include evidence of population health (presence of disease or signs of chronic physical stress) which over time affect population size and demographics.	Population size and demographic structure (buck to doe ratios) directly indicates the viability and health of the population; demographics can affect breeding systems affecting genetic diversity. Densities greater than 15/mi ² can indicate population stress due to over-population (Heffelfinger 2006).	Stressors affecting population condition include quality and availability of forage, presence of infectious disease and decreased water availability.

KEA Class: Name	Definition	Rationale	Stressors
<i>Biotic Condition: Forage Quality</i>	Relative composition, abundance and species richness of native forage plants, average vegetation density and height within habitat areas.	Composition of grasses, forbs, succulents, shrubs and trees indicate forage quality. Coues white-tailed deer require a diversity of vegetation for forage and thrive with higher plant species richness. They are opportunistic herbivores; high-protein, succulent forbs, shrubs and trees make up the majority of their diet, with grasses equaling a small portion of their diet. Availability of both native shrubs and trees is shown to be essential for fawn survival (Ockenfels 1991). Coues white-tailed deer require a diversity of vegetation for forage and thrive with higher plant species richness. They are opportunistic herbivores; high-protein, shrubs make up the majority of their diet, with grasses and other plants making up a minor portion of their diet. Availability of both native shrubs and trees is shown to be essential for fawn survival (Ockenfels 1991). Coues white-tailed deer are associated with vegetation within reach. Access to vegetation ranging from 2.5 inches to approximately 60 inches is necessary. Having access to lower shrubs, forbs and trees is critical for fawns.	Historically, the main stressors for white-tailed deer have been the competition for food with grazing cattle, Hunting mismanagement and loss of habitat (Heffelfinger 2006). During times of drought, livestock management acts as a change agent for white-tailed deer through increased competition for forage and water sources. Grazing allotment boundaries may indicate locations of habitat with direct competition for forage and water from cattle, and possible reduced forage quality. Locations of stock waters are indicators of heavy use of vegetation.
<i>Abiotic Condition: Availability of Surface water</i>	Annual and seasonal precipitation and the resulting density, distribution and distance to surface water accessible by Coues white-tailed deer.	Annual reproductive success in Coues white-tailed deer has shown to depend heavily on winter rainfall. Winter precipitation increases high nutrient forage and surface water resources in spring and early summer. Coues white-tailed deer require regular access to surface water and are usually found within a 3/4-mile radius of an accessible, open water source (Ockenfels 1994).	Changes in precipitation regime and average temperatures may change the availability of surface water. Human use of water, and competition from livestock management will exacerbate this.
<i>Abiotic Condition: Temperature Regime</i>	The pattern of average temperature fluctuations; the duration, intensity and seasonal timing of temperature extremes.	Coues white-tailed deer rely greatly on specific habitat requirements to survive and thrive. These habitat requirements are affected by precipitation, temperature, seasonality, and other climatic conditions. Although Coues white-tailed deer are able to obtain much of the water that they need from their diet, the absence or reduction of succulent forage requires deer to seek out additional surface water.	High temperatures in conjunction with low precipitation can affect forage quantity and quality, surface water availability, and may directly result in fawn and juvenile mortalities due to hyperthermia. Water source congestion during the hottest period of the year can cause overbrowsing. If conditions persist survival may be threatened (Truett 1971). Climate change also has the potential to cause additional change in the hydrologic regime, through its effects on precipitation form (snow vs. rain), spatial distribution, magnitude, and timing; and through its effects of evapotranspiration rates both within the riparian zone and across the surrounding watershed. Climate change may also cause changes in human water use.

D-3.6 Conceptual Model Diagram

A conceptual model diagram for each CE provides a visual summary representation of the Key Ecological Attributes and species' life cycle (Figure D-6). These diagrams are intended to show how various stressors interact with categories of Key Ecological Attributes to highlight important drivers for species ecological integrity. The arrows indicate relationships between stressors and but do not indicate the nature of the influence (i.e. positive or negative).

Figure D-6. Conceptual model for the North American Coues white-tailed deer, showing key ecological attributes (by class) for the Coues white-tailed deer.



D-3.7 References for the CE

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D-4 Pronghorn (*Antilocapra americana*)

D-4.1 Classification Comments

Endemic to North America, pronghorn (*Antilocapra americana*) are the last surviving genus in the family Antilocapridae. Historically debated to be part of the Bovidae family, recent genetic studies have confirmed that they belong in Antilocapridae (Janis and Scott 1987; Jones et al. 1992; Wilson and Reeder 1993, 2005), and are more closely related to Giraffidae than to Cervidae or Bovidae despite their antelope-like appearance (Hernandez et al 2005; Murray 2006; Janis 2000). Pronghorn are unique in having horns made of a keratinous sheath over a bony core, similar to bovids, that they shed and re-grow each year, similar to antlers of cervids (AZGFD 2011).

Five subspecies of the North American pronghorn are currently recognized: the American pronghorn (*A. a. americana*, Ord 1815); the Mexican or Chihuahuan pronghorn (*A. a. mexicana*, Merriam 1901); the Baja California or Peninsular pronghorn (*A. a. peninsularis*, Nelson 1912); the Oregon pronghorn (*A. a. oregona*, Bailey 1932); and the Sonoran pronghorn (*A. a. sonoriensis*, Goldman 1945) (ITIS, accessed 18 January 2013; USFWS 2013; O’Gara and Yoakum 2004). The validity of these subspecies and their separate ranges are not fully understood, and are largely defined by population variations in color, size, and location (O’Gara and Yoakum 2004). Genetic studies of North American pronghorn populations led Lee et al. (1994) and Cockrum (1981) to question subspecies status based on these minimal physical and geographical differences; particularly between the American pronghorn and Chihuahuan pronghorn occurring in the Madrean Archipelago ecoregion and western Texas, and between the American pronghorn and Oregon pronghorn populations in Oregon, California, Idaho, and Nevada (Lee et al. 1994; Lee 1992). These studies and other more recent mitochondrial DNA analyses lend support to the identification of clines, rather than distinct subspecies, for pronghorn populations displaying minor phenotypic or genetic differences that reflect adaption to the diverse habitats and environments in which pronghorn range (O’Gara and Yoakum 2004). The genetic integrity of these taxonomic subspecies are further complicated by translocation and reintroduction efforts that began in the early twentieth century and continue today, in an attempt to bring back declining or extirpated populations in the region (USFWS 2013). Currently, subspecies distinctions remain standard for pronghorn populations throughout North America.

For the Madrean Ecoregional assessment we shall focus on two subspecies, the American pronghorn (*A. a. Americana*, Figure D-7 and Figure D-8) and the Mexican or Chihuahuan pronghorn (*A. a. mexicana*), unless otherwise noted. References to the Sonoran pronghorn (*A. a. sonoriensis*) are occasionally used to understand unique adaptation responses relevant to this arid borderlands ecoregion.

Figure D-7. American pronghorn (*A. a. americana*), Animas Valley, Arizona. Photo © 2012 Sky Island Alliance/ Melanie Emerson.



Figure D-8. American pronghorn (*A. a. americana*), Las Cienegas National Conservation Area, Arizona. Photo © 2012 Sky Island Alliance/ Tim Cook.



D-4.2 Biology and Distribution Status

D-4.2.1 Distribution

North America: Pronghorn once ranged from southern Canada to northern Mexico; and from the Mississippi River and Gulf Coast of Texas to California and the Pacific Ocean. Today pronghorn populations are primarily distributed along the “Spine of the Continent” (Hannibal 2012) in southeastern Oregon, southern Idaho, southern Alberta and Saskatchewan, Montana, and western North Dakota to Arizona, western Texas, and northern Mexico (Byers 1997; O’Gara and Yoakum 2004), and were unsuccessfully introduced to Lanai, Hawaii in 1958 (NatureServe 2007).

Madrean Archipelago ecoregion: Mixed populations of the American and Chihuahuan pronghorn subspecies occur within the Madrean Archipelago ecoregion today (AZGFD 2011; Howard 1995; Lee et al 1994; Yoakum 1980). Historically, Sonoran pronghorn was distributed from southern Arizona to the desert plains of central and western Sonora, Mexico (CONANP 2009; Leopold 1959). The Chihuahuan pronghorn's distribution encompasses southeastern Arizona, southwestern New Mexico, and portions of west Texas, extending south in Mexico to Chihuahua, Durango, Coahuila, portions of Nuevo Leon and Tamaulipas, and as far south as the state of Hidalgo (CONANP, 2009). These populations consist of small herds scattered throughout their range. Within the Madrean Archipelago ecoregion of Arizona and New Mexico, seven population localities are currently managed and monitored (AZGFD 2013). In the New Mexico portion of the Madrean Archipelago ecoregion the Chihuahuan pronghorn occurs in the grassland areas of New Mexico Game Management Units 23, 24, 25, 26, and 27. In 2009, the range of the Chihuahuan pronghorn in northern Mexico principally included localities of Chihuahua, Mexico, including La Perla, La Gregoria, San Luis, Terraceño, El Sueco-Moctezuma, Janos-Ascención and Coyame, as well as Valle de Colombia and Rancho El Novillo near Coahuila, Mexico (CONANP 2009; Pallares 1999). The distribution of pronghorn populations in southeastern Arizona appear to be strongly associated with identified Wildlife Linkage zones in that portion of the region (Arizona Wildlife Linkage Workgroup 2006).

D-4.2.2 Population

Chihuahuan pronghorn subspecies populations occurring in Sonora and Chihuahua have CITES I status and are listed as endangered by the Mexican government; the North American pronghorn is listed by IUCN as Least Concern, Ver. 3.1 (Hoffman et al 2008). Recent aerial pronghorn surveys conducted by New Mexico indicate a declining population within the ecoregion due to low recruitment rates. Arizona's statewide pronghorn population status is based on the sum of population estimates gathered from game management units; these are produced with computer simulation models using data gathered from aerial surveys and annual hunter-reported harvest data. Fixed-wing aircraft surveys are conducted, prior to hunting season, to collect age and sex ratios and population estimates using a simultaneous double count method (AZGFD 2011). An adult pronghorn population of 8,000 was estimated for Arizona (not including Indian reservations), occupying approximately 21,000 square miles of habitat; this population is projected to increase steadily 2 to 4 percent per year over the next ten years (AZGFD 2011). The population trend in Mexico and the southwest United States is in decline, despite the projected upward trend for Arizona, largely due the recent severity of droughts and winters (Hoffman et al 2008; O'Gara 1999). Southern Arizona populations are periodically augmented, and in some cases re-established, with translocated animals from northern Arizona and Texas (AZGFD 2011).

In a series of aerial surveys conducted in 2011, a total of 165 pronghorn were counted from the seven herds managed in the Madrean Archipelago ecoregion (AZGFD 2011); as of 2013 this number is between 286 – 407, despite significant decline due to drought in three herds since 2007 (Caroline Patrick-Birdwell, environmental consultant, pers. communication). Seven pronghorn herd populations are currently managed by the Arizona Game and Fish Department. The most recent 2013 population status for these herds is as follows (Caroline Patrick-Birdwell, environmental consultant, pers. communication):

Altar Valley Population: 11-20 animals (no management goal in the near future; this population has experienced close to zero fawn recruitment in recent years), located in the Altar Valley and Buenos Aires National Wildlife Refuge, near Sasebe and Arivaca, Arizona (within Arizona Hunt Units 36A, 36B and 36C). Population trend is declining.

San Rafael Population: 9 animals (management goal is 100), located in the San Rafael Valley and Fort Huachuca Military Reservation near Lochiel, Arizona and the U.S.-Mexico border (Arizona Hunt Units 35A and 35B). Population trend has been declining since 2007.

Las Ciénegas Population: 60-80 animals (management goal is 50-100), located at Empire Ranch and the Las Ciénegas National Conservation Area northeast of Sonoita, Arizona (Arizona Hunt Unit 34B).

Elgin-Rosetree Population: 19 animals (management goal is 100), located south of the Las Ciénegas Population and bounded on the south by the Huachuca and Patagonia Mountains and Fort Huachuca Military Reservation. Population trend has been declining since 2007.

North Sulphur Springs Valley-Bonita Population: 50-80 animals (management goal is 100-200), located in the Sulphur Springs Valley and Allen Flat area north of Interstate-10 and Dragoon, Arizona and west of Highway 191 (Arizona Hunt Units 31 and 32). Population trend has been declining since 2007.

San Bernardino Valley Population: 125-175 animals (management goal reached), located in the San Bernardino Valley northeast of Douglas, Arizona (Arizona Hunt Unit 30A).

Lordsburg Plains Population: 12-24 animals within Arizona (no management goal) representing an estimated 10% of the total pronghorn population (120-240 animals) on the Lordsburg plains, located in the Day Ranch area of the Animas Valley, spanning across the border of Arizona and New Mexico, east of the Peloncillo Mountains and north of Interstate-10 (Arizona Hunt Unit 28), and also bounded by the Gila River and Burro Mountains (New Mexico Hunt Units 27 and 23). This population moves fluidly between Arizona and New Mexico.

D-4.2.3 Reproduction

Breeding Systems: Male pronghorn will exhibit different breeding systems within the same population and this behavior is a direct result of the availability and location of resources, density of the population, and the ratio of males to females in a group (Maher 2000). Territorial breeding behavior (male defense and competition of a defined location containing water, succulent vegetation and the females within this area) occurs when quality resources are isolated or scarce. Dominant breeding behavior occurs when resources uniformly available on the landscape, resulting in the formation of herds of females and multiple males, with the most dominant males in the hierarchy doing most of the breeding. Harem breeding behavior occurs when resources are widely dispersed, population levels are low, or the sex ratio is skewed (1 male to 10 or more females); in this case, a single breeding male will defend a harem of females without regard to a defined territory.

Gestation: The gestation period for pronghorn is between 245 and 255 days. Ova are able to develop for up to a month after fertilization before implantation occurs (San Diego Zoo Global Library 2009; O'Gara and Yoakum 2004; Byers 1997; O'Gara 1978). Females may reabsorb embryos under conditions of environmental stress.

Litter size: One to two fawns, per breeding female per year. Twins are the norm; occasionally less dominate females will give birth to a single fawn.

Breeding Season: Pronghorn in southern habitats breed in the fall, between the months of September and October. Breeding behavior may begin in late summer, as early as July (San Diego Zoo 2009; Byers

1997; O'Gara 1978; O'Gara & Yoakum 2004). Northern pronghorn populations begin their breeding season earlier.

Birthing Season: Depending on habitat and environmental conditions, pronghorn generally give birth in the Madrean Archipelago ecoregion between June and July; this differs from northern populations that typically give birth in May and June. Pronghorn exhibit synchronous birthing, with gravid females in a herd giving birth within the same 10 day period (San Diego Zoo 2009; Byers 1997; O'Gara 1978; O'Gara & Yoakum 2004).

Life history: Pronghorn weigh 7-9 pounds at birth, which is approximately 18% of the adult female's weight. Young are weaned by 4 to 5 months of age but do not reach their full adult weight for about 4.5 years. Males reach sexual maturity in one year but may not have the opportunity to breed until they are dominant enough to compete with other males. Females are sexually mature at 16 months, but there are cases of females conceiving as young as 5 months (San Diego Zoo 2009; Byers 1997; O'Gara 1978; O'Gara & Yoakum 2004). In the wild, the lifespan of a pronghorn is less than 10 years.

D-4.3 Habitat and Ecology

D-4.3.1 Habitat

Pronghorn have several specific habitat requirements needed for survival. Yoakum et al. (1996) and Jaeger and Fahrig (2004) defined the optimal habitat parameters for the North American pronghorn as elevation, terrain, landscape connectivity, distance from water, and vegetation. In 1995, the Arizona Game and Fish Department conducted a statewide analysis that quantified and ranked pronghorn habitat based on five variables: topographic ruggedness, vegetative structure and species richness, water availability, human disturbance, and fence density and structure (Ockenfels et al. 1996; Ockenfels et al. 2000).

Elevation: In Arizona and New Mexico pronghorn are found between 3,000-7,000 feet (914-2,134 meters) elevation, although northern Arizona herds have been documented as high as 10,000 feet (3,048 meters) in summer (AZGFD 2011). New Mexico has also documented several pronghorn populations at high elevations above 8,000 feet (2,438 meters) as high as 10,000 feet (3,048 meters).

Terrain: Yoakum et al. (1996) cites that pronghorn prefer open habitat, with flat or with rolling hills, in order to detect and avoid or escape approaching predators; pronghorn are associated with topographic ruggedness of less than 20% slope according to studies in Arizona (AZGFD 2011; Ockenfels et al. 1996; Ockenfels et al. 2000).

Distance from accessible water: Pronghorns require regular access to surface water and are usually found within a 4 mile (6.5 km) radius of an open water source (Yoakum et al, 1996). These include but are not limited to natural springs and seeps, human-made wildlife waters and cattle tanks, ponds, lakes, and perennial riparian corridors and streams.

Vegetation: Largely associated with grasslands, short-grass prairie, and shrub-steppe, preferably with vegetation ranging 5 to 30 in (13 to 76 cm) in height (Byers 1997, O'Gara & Yoakum 2004). Pronghorn are occasionally found in temperate desert habitat; deserts support less than 1% of the North American pronghorn population, and this consists largely of the Sonoran subspecies (San Diego Zoo 2009, Byers 1997, O'Gara & Yoakum 2004). Based on population distribution data from the Arizona Game and Fish Department (AZGFD 2013) and according to the biotic communities described in Brown and Lowe (1982), pronghorn herds in the Madrean Archipelago ecoregion occur primarily in semi-desert

grasslands and Plains and Great Plains grasslands, with the Sulphur Springs Valley population also overlapping the edges and open areas of Madrean oak woodland habitat.

D-4.3.2 Landscape connectivity

Pronghorn have become the poster child for landscape connectivity and large-scale wildlife corridor conservation (Hannibal 2012), as they require expansive areas and connected habitats to thrive. One population in the Wyoming Greater Yellowstone region is known for having the longest seasonal migratory pathway of any other terrestrial species in the continental U.S., and using the same 100 mile (160 km) route for more than 6,000 years (Hannibal 2012, San Diego Zoo 2009, Byers 1997, O’Gara & Yoakum 2004). In the Madrean Archipelago region these seasonal migrations are not as impressive and are limited within separate herd home ranges, but are essential to maintaining gene flow and finding quality forage and water sources in summer and winter. Barriers to movement result from fragmentation and habitat loss caused by human development, in the form of fencing, mining, urban sprawl, and roads, railroads and highways, among others (AZGFD 2011). In the Madrean Archipelago ecoregion, major barriers include the U.S. – Mexico border and Interstate-10. New Mexico Department of Game and Fish has partnered with the US Forest Service to identify and implement habitat projects designed to enhance historical pronghorn corridors between summer and winter range that has pinyon/juniper encroachment.

Pronghorn cross fences by passing underneath, rather than jumping over; barbed wire, woven wire or fences with the bottom wire lower than 20 inches (< 50cm) off the ground are barriers for pronghorn (Yoakum et al. 1996, Jaeger and Fahrig 2004). Fences become even greater barriers to pronghorn movement (and gene flow due to direct mortality) when they are placed too close to high-traffic roads or highways without a buffer zone that allows individuals to navigate one obstacle at a time (AZGFD 2011; Scott Sprague, AZGFD pers. communication). Pronghorn-friendly fencing must have a smooth bottom wire at least 20 inches above ground level, or a bottom wire equipped with plastic pipe “goat bars” (AZGFD 2011).

D-4.3.3 Food

Pronghorn require a diversity of vegetation for forage and thrive with higher plant species richness. They are opportunistic herbivores; high-protein, succulent forbs make up the majority of their diet, along with shrubs (i.e. rabbit bush, sagebrush), grasses (i.e. bunchgrass, squirrel-tail) and cacti (*Opuntia* sp.). Grasses total approximately 12% of their diet (AZGFD 2011; Yoakum et al. 1996; O’Gara 1978). Availability of both grasses and shrubs is shown to be essential for fawn survival (AZGFD 2011). Pronghorn frequently change their forage preferences season to season, and will seek the most nutritious and succulent forage species available (San Diego Zoo 2009, Byers 1997, Hansen & Clark 1977, O’Gara 1978, O’Gara & Yoakum 2004). In harsh winters, northern Arizona populations had a higher survival rate with more shrub forage available than herds who fed largely on grass (AZGFD 2011).

D-4.3.4 Phenology

Adult and juvenile pronghorn activity patterns are circadian, crepuscular and diurnal (NatureServe 2007, Einarsen 1948, Kitchen 1974). Pronghorn have crepuscular feeding habits, with most foraging activity occurring at dawn and dusk; however, the majority of their time throughout the day is spent foraging, ruminating and sleeping or resting. Approximately 40 to 60 percent of their day is spent feeding, and they sleep without regularity and for frequent, short periods (San Diego Zoo 2009, Byers 1997, Hansen & Clark 1977, O’Gara 1978, O’Gara & Yoakum 2004). Activity patterns for pronghorn are variable, and influenced by region, weather, season, and also by herd dynamics. Some pronghorn populations migrate long distances between summer and winter feeding grounds, although this does not appear to be the case with southern herds that occur in warmer habitats where quality forage is available year-round.

Northern herds have shown daily foraging movements that vary greatly, from 0.1 to 0.8 km in the spring and summer to 3.2 to 9.7 km in the fall and winter. In the Madrean ecoregion, home ranges are anywhere between 20 to 40 square miles (AZGFD 2011), and herds travel seasonally between fawning grounds and winter and summer feeding grounds within each home range (Caroline Patrick-Birdwell, environmental consultant, pers. communication).

D-4.3.5 Predators

Coyotes (*Canis latrans*), bobcats (*Lynx rufus*), mountain lions (*Puma concolor*), golden eagles (*Aquila chrysaetos*), gray wolves (*Canis lupus*), and humans are all predators of the pronghorn; the only other predator it evolved with is the jaguar (*Panthera onca*). Fawn predation is of most concern; due to their synchronous birthing behavior, small populations may lose the majority of fawns to heavy predation occurring in a short time period (AZGFD 2011).

D-4.3.6 Competitors

Pronghorn compete with other browsers and grazers on the landscape. O’Gara (1978) calculates that 38 pronghorn can survive on the same amount of forage that is needed for a single cow. Research on the Sonoran pronghorn suggests that livestock grazing can be a significant stressor on pronghorn populations in combination with poor forage or drought conditions (USFWS 2002).

D-4.3.7 Demographics

Sparse or small populations can result in random variations in sex ratios, age distributions, and birth and death rates among individuals, which can cause fluctuations in population size and even lead to extinction (USFWS 2002). In very small isolated populations, males may have trouble finding females, leading to decreased recruitment (USFWS 2002). The composition of a herd can change hourly or day to day, forming loosely dispersed groups of 3 to 25 that keep in visual contact when food is readily available, or forming large herds of 1,000 individuals that are made up of many groups that band together, most often in fall and winter in northern areas where predation is higher (Byers 1997, O’Gara 1990, O’Gara & Yoakum 2004). Dominance hierarchies are maintained by female bands, bachelor bands, and territorial male bands that contain more than one male (San Diego Zoo 2009).

D-4.3.8 Disease and parasites

Pronghorn can suffer from hemorrhagic disease caused by epizootic hemorrhagic disease viruses and the Bluetongue virus, also known as catarrhal fever, which is a virus transmitted by *Culicoides* sp. during warm summer months. In Wyoming in 1976, Bluetongue virus killed nearly 3,200 pronghorn; and another 300 mortalities occurred in 1984 (Thorne et al. 1988). *Haemonchus contortus* a stomach worm of sheep and cattle has impacted pronghorn in Texas (McGhee 1981, Wasel 2003). White-tail deer (*Odocoileus virginianus*), mule deer (*Odocoileus hemionus*), and bighorn sheep (*Ovis canadensis*) act as primary, unaffected, hosts that pass it on to other species through their feces; pronghorn ingest vegetation with gastropods, the intermediate host, that have become infected with the larvae of the nematode (Simmons et al. 2002). Hemorrhagic disease has been identified occasionally in free-ranging white-tail deer, mule deer and bighorn sheep in Arizona (Dubay et al. 2006).

D-4.4 Change Agent Characterization

This section of the conceptual model presents a narrative description of the primary change agents and current knowledge of their effects on Pronghorn populations. The main change agents identified are forms of altered dynamics: Livestock management, development, and climate change.

D-4.4.1 Livestock management

Competition for food with grazing cattle is one stressor on pronghorn, most significantly during poor forage conditions, and heavy livestock use coupled with frequent drought periods reduces forbs during the growing season (AZGFD 2011, USFWS 2002). Livestock management acts as a change agent for pronghorn through increased competition for forage and water sources, and increased fencing, cattle guards, and other barriers on the landscape. Livestock management can provide additional water resources available for pronghorn, but only if it is not at cost to water availability (Yoakum et al. 1995) and wildlife friendly fencing may be used. In Mexico, where invasive forage such as buffelgrass (*Pennisetum ciliare*) has been subsidized and planted for livestock grazing, decreased plant diversity and changing fire regimes have detrimental effect. Loss of habitat and movement corridors from fences used to control livestock movements within pastures, between ranches and allotments, and along highways, and habitat degradation from long-term vegetative community changes due to livestock overuse and fire suppression greatly impact pronghorn populations (AZGFD 2011; USFWS 2002; Ockenfels et al. 1994). Cooperative livestock management practices can provide benefits and reduce stressors on pronghorn. New Mexico BLM has utilized various tools to initiate the restoration process within many of the desert grasslands across the southern portions of the state. This restoration process has included the reduction of species that have invaded these grasslands, including creosote and mesquite. This statewide initiative, known as Restore New Mexico, will culminate in the restoration of many desert grassland habitats which are home to pronghorn.

Stresses: increased competition, increased habitat fragmentation, change in vegetation structure, decreased plant diversity, decreased forage abundance, decreased water quality, increased soil compaction, habitat degradation, increased behavioral disruption, increased fire frequency and intensity.

Responses: decreased population size, reduced home range size, reduced migratory distance, reduced recruitment, increased mortality rate.

D-4.4.2 Development

Human development influences pronghorn populations by directly affecting habitat and resources, and may fall into the categories of 1) Military and Border-defense related activity; 2) Mining; 3) Rural and urban development including canals; 4) Transportation in the form of roads, railroads and highways; and 5) Energy Development including new transmission lines, and converting open space for solar and wind farming. The U.S. Fish and Wildlife Service (2011) cites that the difference between rates of decline north and south of the border for Sonoran pronghorn populations might be due to high levels of human disturbance on the U.S. side, from illegal immigration, smuggling, and law enforcement response to these activities (USFWS 2011); these stressors are also present in the Madrean Archipelago ecoregion where pronghorn occur. Border infrastructure, including lighting, roadways, and frequent vehicle traffic, pose a significant barrier to wildlife movement north and south of the border, and large scale mining can have a tremendous impact on ground water resources, water and air quality, and habitat availability. Rural and urban develop has a direct impact on ground water levels and habitat resources, and transportation and energy development results in habitat fragmentation and direct mortality from vehicle collisions. Increased human development also infers increased human presence: increased access to more remote areas generated by increased road development may increase direct take from legal and illegal harvest, incidental road kill, and other disturbance. Brown et al (2012) determined that elk and pronghorn did not respond to noise levels associated with motorized human activity, which they suggested was due to the animal's inability to maintain constant responsiveness to these activities.

Brown et al also suggested that although less responsiveness may be an investment in fitness, it could potentially decrease predator detection and increase human conflicts.

Stresses: reduced habitat/ open space, decreased water availability, increased habitat fragmentation, decreased water quality, habitat degradation, increased behavioral disruption.

Responses: decreased population size, reduced recruitment, decreased home range, reduced migratory distance, decreased gene flow/ decreased genetic health, increased mortality rate.

A note on the potential side effects of pronghorn management including sound disturbance: translocation of animals from other herds to augment populations with the goal of stabilizing populations and increasing hunting recreation opportunities (AZGFD 2011), leads to increasing hybridization between subspecies and occasional capture myopathy (mortality due to capture). In addition, pronghorn are routinely pursued by helicopter or fixed-wing aircraft for capture efforts and annual population counts, which may increase stress in the form of behavioral disruption, predator response, herd dispersal and occasional physical injury.

D-4.4.3 Climate change

Pronghorn rely greatly on specific habitat requirements to survive and thrive, which are affected by precipitation, temperature, and other climatic conditions and change through the seasons. Additional studies of these influences in consideration of a changing climate—including increasing temperature extremes, increased fire intensity and frequency, and drought—are needed, but some research outside the Madrean Archipelago ecoregion may provide insight to the responses that may be expected. Annual reproductive success in Sonoran Pronghorn has shown to depend heavily on winter rainfall (Heinz Center 2011), presumably because winter precipitation increases high nutrient forage and surface water resources in spring, a factor that also occurs in the similarly arid Madrean ecoregion. Depending on conditions, pronghorn are able to obtain some of the water that they need from their diet, but many pronghorn populations use water sources, particularly during dry periods. In the absence or reduction of succulent forage pronghorn have been recorded using water developments (O’Gara and Yoakum 1992; Yoakum 1994). In desert environments, Sonoran Pronghorn, previously thought not to use free water have been detected drinking at water sites (Morgart et al. 2005). Free water does influence the distribution of pronghorn. In Arizona Sonoran pronghorns were typically found to be within 2-6 km of an open water source (Hughes and Smith 1990; Devos and Miller 2005) and fawns were found to bed within .4-.8 km of water (Ticer and Miller 1994). In other parts of the country, pronghorns have been shown to stop using water sources when forage moisture content exceeds 75 percent (Beale and Smith 1970). Decreased water resources and increased presence of predators at available water sites can expose pronghorn to greater risk of mortality; in addition, warmer temperatures and lower precipitation may lead to increased water drawdowns by human populations (Heinz Center 2011, SWCCN 2008). Some studies (Wilson and Krausman 2008) have speculated that fawns and juveniles may be susceptible to hyperthermia during the hottest period of the year (for temperatures reaching 109 °F (43 °C) or greater), leading to death. Extreme cold temperatures in combination with low forage quality and diversity can also cause direct mortality to pronghorn (AZGFD 2011). The National Drought Mitigation Center (2013) reports that the Madrean Archipelago ecoregion has been experiencing Severe to Extreme drought in the last year, and forecasts this condition will persist or intensify.

Stresses: increased change in vegetation structure, change in biotic community, decreased forage availability, decreased water availability, increased distance to water, increased predation, increased fire frequency and intensity, hyperthermia, hypothermia.

Responses: decreased population size, reduced recruitment, increased reliance on artificial water, increased daily foraging movements, increased migratory distance, increased mortality rate.

D-4.5 Ecological Status: Key Ecological Attributes and Indicators

This section of the conceptual model addresses Key Ecological Attributes and their potential indicators. The ecological status is a way of describing current status via criteria, functionality, or levels of attributes and asks if they are within the normal range of variation.

D-4.5.1 Key Ecological Attributes

Table D-5 identifies the key ecological attributes for this CE within the Madrean Archipelago ecoregion. A **key ecological attribute** of a focal ecological resource is a characteristic of the resource's biology, ecology, or physical environment that is critical to the resource's persistence in the face of both natural and human-caused disturbance. Alteration of such a characteristic beyond some critical range of variation will lead to the degradation or loss of the resource within decades or less. The KEAs table lists the identified key ecological attributes, with a brief definition, a rationale for why it is important for the CE, and a listing of stressors or change agents that might be affecting the key attribute.

Table D-5. Key Ecological Attributes (KEA) used to determine the ecological status of North American Pronghorn (*Antilocapra americana Mexicana* and *A. a. Americana*) occurring in the Madrean Archipelago ecoregion.

KEA Class: Name	Definition	Rationale	Stressors
Landscape Context: Landscape and Habitat Condition	The extent of suitable grasslands, short-grass prairie and shrub-steppe that is intact and unfragmented.	Pronghorn have specific habitation requirements for survival including grasslands with low topographic ruggedness between 3,000—7,000 feet elevation and regular access to surface water. Pronghorn habitat and historical herd home ranges and dispersal movements extend north and south of the U.S.-Mexico border; pronghorn populations occurring in the Madrean ecoregion are directly affected by the physical barriers to gene flow, behavior disturbance, and habitat loss and destruction, among other stressors.	Habitat fragmentation is caused by livestock management (fences, cattle guards, and roads), mining, rural and urban development, transportation features including roads and railroads, energy transmission lines, canals and border infrastructure. Pronghorn habitat and historical herd home ranges and dispersal movements extend north and south of the U.S.-Mexico border; pronghorn populations occurring in the Madrean ecoregion are directly affected by the physical barriers to gene flow, behavior disturbance, and habitat loss and destruction, among other stressors. This can decrease pronghorn home range, recruitment, disrupt migration, decrease gene flow and genetic health an increase mortality.
Landscape Context: Habitat and Corridor Permeability	The permeability of the landscape to pronghorn movement based on the absence of barriers such as fences, roads and other developments.	Landscape permeability is a measure of connected habitat that can be modeled at various scales: at a fine/ local scale, permeability is critical to pronghorn ability to avoid predators and access food and water; at a larger landscape scale, it is critical for pronghorn population health through breeding between herds, dispersal, and seasonal migration. Daily and seasonal migratory movement between habitat patches is essential for pronghorn adaptability to changing forage and water availability, predator avoidance, and gene flow (AZGFD 2011). Daily and seasonal migratory movement between habitat patches is essential for pronghorn adaptability to changing forage and water availability, predator avoidance, and gene flow (AZGFD 2011). Daily and seasonal migratory movement between habitat patches is essential for pronghorn adaptability to changing forage and water availability, predator avoidance, and gene flow (AZGFD 2011).	Barbed wire, woven wire or fences with the bottom wire lower than 20 inches (< 50cm) off the ground are significant barriers for pronghorn movement (Yoakum et al. 1996, Jaeger and Fahrig 2004). Fences also cause direct mortality when placed too close to high-traffic roads or highways without a buffer zone that allow individuals to navigate one obstacle at a time (AZGFD 2011; Scott Sprague, AZGFD pers. communication). Pronghorn fencing must have a smooth bottom wire at least 20 inches above ground level, or a bottom wire equipped with plastic pipe "goat bars" (AZGFD 2011). Fencing that is not permeable for pronghorn restricts available habitat for viable populations to find forage, water, mates and to avoid predators, and influences gene flow and migratory behavior. Low or absent landscape permeability can also indicate locations of corridor bottlenecks and/or genetic isolation between populations that can lead to decreased genetic health, inbreeding and local extinction. This indicator can include a number of landscape or habitat features that decrease or alter movement patterns of the CE- fences, rough topography, roads, or other barriers to movement.

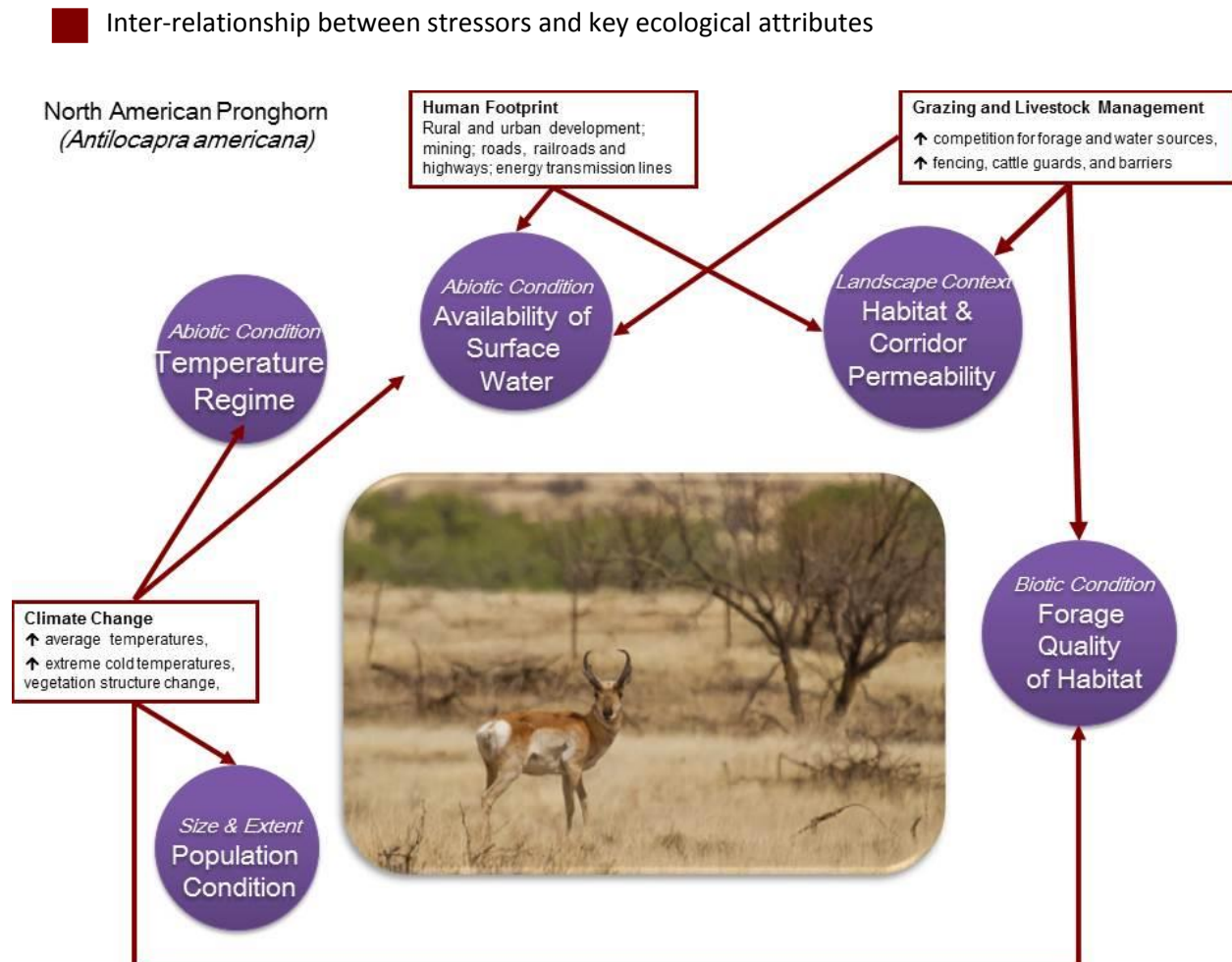
KEA Class: Name	Definition	Rationale	Stressors
Size/Extent: Population Condition	The size and age and sex distribution of pronghorn populations.	Population size and demographic structure directly indicates the viability and health of the population; demographics can affect breeding systems affecting genetic diversity.	Human development may impact population size and demography by reducing suitable habitat, direct disturbance and migration disruption. Other stressors include reduced habitat quality, direct disturbance, mortality due to indirect effects of climate change and impacts from livestock grazing management. Fawns and juveniles may be susceptible to hyperthermia during the hottest period of the year which can cause direct mortality.
Biotic Condition: Quality and Availability forage	The amount and quality of forage available to pronghorn within suitable habitat based on native species composition, vegetation height and amount of invasive ground cover.	Composition of grasses, forbs, succulents and shrubs indicate forage quality and rangeland health. Pronghorn require a diversity of vegetation for forage and thrive with higher plant species richness. They are opportunistic herbivores; high-protein, succulent forbs make up the majority of their diet, with grasses equaling 12% of their diet (AZGFD 2011; Yoakum et al. 1996; O’Gara 1978). Pronghorn are associated with vegetation ranging 5 to 30 in (13 to 76 cm) in height (Byers 1997, O’Gara & Yoakum 2004). Availability of both native grasses and shrubs is shown to be essential for fawn survival (AZGFD 2011). O’Gara (1978) calculates that 38 pronghorn can survive on the same amount of forage that is needed for a single cow; this makes livestock grazing a significant stressor on pronghorn populations, particularly in poor forage conditions (USFWS 2002).	Historically, the main stressor for pronghorn has been the competition for food with grazing cattle (USFWS 2002). Livestock management acts as a change agent for pronghorn through increased competition for forage and water sources, and increased fencing, cattle guards, and other barriers on the landscape. Grazing allotment /pasture boundaries may indicate locations of some impermeable fencing types; additionally, active grazing leases can indicate habitat with direct competition for forage and water from cattle, and possible reduced forage quality.
Abiotic Condition: Availability of Surface Water	The availability of surface water for drinking and to support forage as measured by annual and seasonal precipitation and the density of surface water resources.	Precipitation affects surface water availability and growth of high-nutrient forage. Annual reproductive success in Sonoran Pronghorn has shown to depend heavily on winter rainfall (Heinz Center 2011), presumably because winter precipitation increases high nutrient forage and surface water resources in spring. Pronghorns require regular access to surface water and are usually found within a 4 mile (6.5 km) radius of an accessible, open water source (Yoakum et al, 1996).	Decreased water resources and increased presence of predators at available water sites can expose pronghorn to greater risk of mortality. Providing livestock water in habitat that is already suitable to pronghorn in terms of water distribution (3-4 miles distant) will negatively impact pronghorn in terms of reduced availability of quality forage (Yoakum et al. 1995) Decreases in high nutrient forage and surface water resources will stress pronghorn populations forcing them to move more to forage and reach open water and potentially increasing mortality.

KEA Class: Name	Definition	Rationale	Stressors
<i>Abiotic Condition: Temperature Regime</i>	The pattern of average temperature fluctuations; the duration, intensity and seasonal timing of temperature extremes.	High temperatures in conjunction with low precipitation can result in greater fire risk (equating to forage and/or habitat loss); in addition, extreme temperatures can affect forage quantity and quality, surface water availability, and may directly result in fawn and juvenile mortalities due to hyperthermia (Wilson and Krausman 2008) during the hottest period of the year if temperatures reach 109 °F (43 °C) or greater. Extreme cold temperatures in combination with low forage quality and diversity can cause direct mortality to pronghorn (AZGFD 2011).	Stressors affecting the temperature regime include riparian corridor development; and alterations to the riparian floral assemblage including invasions of non-native flora with high water consumption. Climate change also has the potential to cause additional change in the hydrologic regime, through its effects on precipitation form (snow vs. rain), spatial distribution, magnitude, and timing; and through its effects of evapotranspiration rates both within the riparian zone and across the surrounding watershed. Climate change may also cause changes in human water use.

D-4.6 Conceptual Model Diagram

A conceptual model diagram for the CE provides a visual summary representation of the Key Ecological Attributes and species' life cycle (Figure D-9). These diagrams are intended to show how various stressors interact with categories of Key Ecological Attributes to highlight important drivers for species ecological integrity. The arrows indicate relationships between stressors and but do not indicate the nature of the influence (i.e. positive or negative).

Figure D-9. Conceptual model for the North American Pronghorn, showing key ecological attributes (by class) for this species and relationship of stressors to KEAs.



D-4.7 References for the CE

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Reptiles

D-5 Desert Box Turtle (*Terrapene ornata* subspecies *luteola*)

D-5.1 Classification Comments

The box turtle (*Terrapene ornata luteola*, Figure D- 10) is a genus of four species and 12 taxa native to North America. Box turtles are terrestrial members of the American pond turtle family (Emydidae). The eastern box turtle (*T. carolina*) with six extant subspecies (*T. c. baueri*, *T. c. carolina*, *T. c. major*, *T. c. mexicana*, *T. c. trianguis*, and *T. yucatana*) and a larger extinct Pleistocene subspecies (*T. c. putnami*) mostly lives in wooded habitats in the eastern United States and eastern Mexico. The Coahuilan box turtle (*T. coahuila*) is an endangered species endemic to the Cuatro Ciénegas Basin in Coahuila, northeastern Mexico. This species has readapted to living in water with a life style similar to mud turtles (*Kinosternon*). The spotted box turtle (*T. nelsoni*) with two subspecies (*T. n. klauberi* and *T. n. nelsoni*) is a tropical species in the Sierra Madre Occidental and tropical lowlands of western Mexico. The ornate box turtle (*T. ornata*) is a grassland animal in the central and southwestern United States and adjacent states in Mexico. The nominate subspecies (*T. o. ornata*) lives in the central United States from western Indiana, to eastern Texas and into Louisiana. The Desert Box Turtle (*T. o. luteola*) inhabits the driest habitats of all box turtles in western Texas, New Mexico, Arizona, and Chihuahua and Sonora, Mexico.

Figure D- 10. Desert Box Turtle (*Terrapene ornata luteola*), near Patagonia, Arizona. Photos by James C. Rorabaugh.



Species description - The ornate box turtle is a small terrestrial turtle, males are usually smaller than females (Redder et al. 2006). The carapace varies in color from black to dark gray to reddish brown, with yellow lines radiating from the center of every pleural scute. Central scutes have yellow dashes and form a discontinuous mid-dorsal line. In the desert box turtle (desert box turtle), the colors of the mature turtles are lighter and more muted than the colors of the young. The yellow radiating stripes on the shell of males disappear with age, but are generally retained in females. The carapace has an oval outline and a domed and dorsally flattened shape resembling a box, with little or no mid-dorsal keel. The plastron is

hinged between the hyoplastral and hypoplastral bones (at the joint between the pectoral and abdominal scutes; Dodd 2001, Stebbins 2003), and it can be completely closed against the carapace, allowing box turtles to completely withdraw their head and feet. All plastral scutes have yellow streaks. The skin is brown with yellow spots.

Terrapene ornata luteola has 10-16 yellow streaks on pleural scutes as opposed to 5-9 in *T. o. ornata* (Ward 1978). The carapace ground color is lighter in desert box turtles, and fades to a uniformly straw- or horn-colored in older animals. The male has a bluish head and red or orange eyes, while females have brown or yellow irises. The shell is up to 5.8 inches (146 mm) long (Brennan and Holycross 2006). There are usually four toes on each hind foot. Males have an enlarged, recurved inner claw on the hind feet that is used during copulation. Reproductive-age males have a concave plastron (females are flat or convex), a red iris (females are typically yellow), a more posterior cloacal opening compared to females, a longer and thicker tail, and red coloration on the legs and sometimes on the jaw. Young desert box turtles have the vertebral stripe more pronounced than the yellow scute streaks. The carapace after hatching is nearly flat and circular (Legler 1960). Rings on the scute margins appear to be added with each growing season, allowing for reasonable age estimation up to about age 14 (St. Clair 1998).

D-5.2 Protection Status

Box turtle populations throughout the United States are thought to be in decline due to human-influenced landscape changes. Because of this concern, the Arizona Game and Fish Department's Turtles Project is initiating monitoring efforts in desert box turtle habitat to determine baseline population levels and to develop management plans for the species. In 2009 the Turtles Project launched the citizen scientist based Ornate Box Turtle Watch program where members of the public are encouraged to report box turtle observations in Arizona (www.azgfd.gov/boxtrutlewatch). The desert box turtle has rarely been seen in Sonora in the last 20 years; a study of its status is urgently needed.

Terrapene ornata is included in CITES Appendix II, which are species that are not necessarily now threatened with extinction but that may become so unless international trade in them is closely controlled. This turtle is protected under a variety of US laws and regulations (review by Dodd 2001). Both *T. ornata* and *T. nelsoni* have *Protegida* status under the Mexican endangered species law, Norma Oficial Mexicana, NOM-059-SEMARNAT-2010 (Diario Oficial de la Federación 2010). In January 2005, the Arizona Game and Fish Department closed the collecting season on desert box turtles in Arizona (Arizona Commission Order 43). Possession of ornate box turtles, *Terrapene ornata*, is prohibited, except for those legally held prior to season closure. The desert box turtle is designated as a Priority Species for Pima County, Arizona in the Sonoran Desert Conservation Plan (Pima County 2013).

D-5.3 Biology and Distribution Status

D-5.3.1 Distribution

Terrapene ornata luteola ranges throughout southeastern Arizona, much of New Mexico, southwestern Texas, and south into northeastern Sonora and much of Chihuahua, Mexico (Stebbins 2003, van Dijk and Hammerson 2011). In Arizona, this turtle is found in the southeastern counties of Cochise, Graham, Pima, and Santa Cruz counties (Brennan 2013, Brennan and Holycross 2006) and Greenlee and Gila (AZGFD, HDMS, 2013). The westernmost desert box turtle populations are in Las Ciénegas National Conservation Area, in the Santa Cruz River near Sahuarita, and Altar Valley in Pima County (Pima County 2013). The southernmost records for desert box turtle in Sonora are near Arizpe in the Río Sonora Valley (30°20'N) and at Nacozari de García (30°19'N), 110 to 115 km south of the Arizona border. Thornscrub box turtle (*T. n. klauberi*) is found as far north as about 29°50'N in east-central Sonora, about 60 km south of Nacozari.

D-5.3.2 Reproduction

Male desert box turtles can reproduce after they reach 7-8 years of age, and females at 8-9 years (Dawson 1998, Brennan 2013, Brennan and Holycross 2006). Courtship and mating usually take place in summer monsoon season in July-August. Female box turtles can retain the sperm after mating for several years. After mating, females lay an average clutch of 2-3 eggs (up to 8 maximum) during the monsoon season, and incubation lasts about seventy days. Hatchlings are dark in color with a yellow stripe down the center of the shell. They can live 25 years or more.

D-5.4 Habitat and Ecology

D-5.4.1 Habitat

In Arizona, desert box turtles mainly inhabit desert grassland, but can also be found in Chihuahuan desertscrub, Madrean evergreen woodland, and Sonoran desertscrub up to 7100 feet (2,165m) in elevation. They require loose soil for burrowing (Hall and Steidel 2003). The southernmost populations near Arizpe are in foothills thornscrub or riparian habitats along the Río Sonora or in oak woodland/grassland at Nacozari de García. Thornscrub box turtle (*T. n. klauberi*) lives in tropical deciduous forest, foothills thornscrub, and oak woodland in Sonora as far north as about 29°50'N in east-central Sonora. The ranges of the two species are separated by about 60 km south.

Elevation: In Arizona, the desert box turtle mostly occurs at 3000-6500 ft (915-2165 m) elevation, but has been found at 2000 ft (610 m) and 7100 ft (2165 m; Brennan 2013, Brennan and Holycross 2006). In Sonora, it has been found at 2526-5110 ft (868-1558 m) elevation.

D-5.4.2 Food

The desert box turtle is omnivorous, feeding on beetles, crickets, grasshoppers, other insects, worms, crayfish, reptiles, eggs, carrion, cactus fruit, cactus pads, berries, and other plant material (Brennan 2013, Brennan and Holycross 2006). They have been seen in shallow temporary ponds in the summer rainy season feeding on tadpoles and aquatic insects. The box turtle often searches under cow dung for beetles and other insects (Legler 1960, Ernst et al. 1994, Stebbins 2003). Considering the American bison (*Bison bison*) ranged into southeastern Arizona as late as a few hundred years ago (Parmenter and Van Devender 1995) and that many large herbivores went extinct at the end of the Wisconsin glacial age 11,000 years ago, dung beetle food resources for desert box turtle were important in the past (Moodie and Van Devender 1978).

D-5.4.3 Phenology

In southeastern Arizona, the desert box turtle takes refuge in subterranean mammal burrows, primarily those of the Banner-tailed kangaroo rat (*Dipodomys spectabilis*), from November through June (Plummer 2004). In New Mexico, the box turtle is associated with black-tailed prairie dog (*Cynomys ludovicianus*) towns (New Mexico Game and Fish Department 1997). Plummer (2004) found that in the fall, entrance into hibernation varied among individual turtles and between years from 18 October to 6 December. Mean daily body temperatures at the beginning of hibernation were about 8°C. From December through February, body temperatures averaged about 9°C and were slightly less than and highly correlated with the 30-cm soil temperature. By April, body temperatures of turtles approached levels exhibited by surface-active turtles (minimum approximately 18°C for moving turtles) but turtles did not emerge from burrows until the summer monsoons began about three months later. Thus, the five-month hibernation period of *T. o. luteola* was contiguous with a three-month estivation period for a total annual period of subterranean refuge of about eight months.

D-5.4.4 Competitors

The desert box turtle is omnivorous, eating a wide variety of plant and animal matter. Other omnivorous species in the desert grassland that feed on insects, fruits and small vertebrates include Sonoran mud turtles (*Kinosternon sonoriense*), Yellow mud turtles (*K. flavescens*), gray fox (*Urocyon cinereoargenteus*), raccoon (*Procyon lotor*), ringtails (*Bassariscus astutus*), coati (*Nasua nasua*) and skunks (*Mephitis* spp.).

D-5.5 Change Agent Characteristics

D-5.5.1 Landscape Modification

Landscape level changes in desert grassland, such as large-scale urbanization and conversion of grassland to agricultural fields, have major impacts on desert box turtles populations. Grazing does not have a significant impact on the desert box turtle, heavier grazing resulting in more open areas likely favors kangaroo rats and more preferred hibernation sites.

Historically, the desert box turtle was reasonably well distributed in desert grassland in southeastern Arizona. Desert box turtle's are often seen on roads, especially during the monsoon season. Hall and Steidl (2003) found that box turtle abundance was lower in high traffic areas, and that mortality was higher on paved roads. In the Madrean Archipelago Ecoregion there has been an increase in vehicle traffic related to border enforcement activities. The amount and speed of vehicle traffic has increased. Roads have been upgraded, experience more frequent vehicular traffic and in some cases are drug daily to clear the road surface to record fresh tracks.

Stresses: habitat loss due to urbanization or agriculture; animals killed on roads by vehicles; animals removed from populations for pets.

Responses: loss of populations; population health and integrity of gene pool damaged by removal of adults killed on roads or removed as pets.

D-5.5.2 Collection of wild animals

Substantial numbers of box turtles have been collected in the past for the domestic and international pet trade, with potentially significant population impacts (van Dijk and Hammerson 2011). Box turtles of several species and varieties are commonly kept as pets in the United States and offered for sale online (The Turtle Shack 2013, The Turtle Source 2013).

Although collection of desert box turtles is prohibited by the Arizona Game and Fish Department, individuals encountered crossing roads in suburban and rural areas in southeastern Arizona may be taken home as pets. In Willcox, Arizona there is an annual box turtle race for which people likely collect wild turtles from local roads. Although the eastern ornate box turtle is commonly sold as pets for as much as \$250 apiece (the Turtle Shack 2013), the desert box turtle is not offered for sale online.

Stresses: increased human presence, animals removed from populations for pets.

Responses: loss of populations; population health and integrity of gene pool damaged by removal of adults killed on roads or removed as pets.

D-5.5.3 Climate Change

Climate change is an ongoing process in the Southwest with potential effects on all species and biotic associations (Thompson et al. 1997). Mean annual temperatures rose by 2.0-3.1°F in the American Southwest in the 20th century, and are predicted to rise between 8.1 and 11.0°F in the 21st century (Southwest Regional Assessment Group 2000). Changes in precipitation are more speculative, with contradicting predictions in different models (Southwest Regional Assessment Group 2000, Patterson

1997, Betancourt 2004). The effects of different climatic change scenarios in my view are discussed here.

Contradictory predictions about future precipitation regimes make evaluating threats to the desert box turtle difficult. With increased annual temperatures, desert box turtle would likely enter into hibernation later in the fall, shortening the hibernation-estivation inactivity period. With increases both in summer precipitation and fall tropical storms, general conditions during the activity period from July to November would be less stressful with increased food resources (insects, fruits, etc.), increased survivorship at all life stages, and potentially more successful dispersal and gene flow between metapopulations.

In desert grassland, shrub dominance has increased as perennial grass and herbaceous perennial dominance declined twice in the last 4,000 years during warm periods before European arrival (Van Devender 1995). Desert box turtle would likely not be impacted significantly by shrub increases in desert grassland. Moodie and Van Devender (1978) reported fossil box turtles from Cochise County, Arizona, including a Pliocene *Terrapene* cf. *ornata* in Pliocene (ca. 4.0 mya [million years ago]) sediments and *T. ornata* in late Pleistocene (ca. 0.5-1.0 mya) deposits. Considering that desert box turtle has survived in the Madrean Archipelago region during frequent climate and vegetation changes for the last two million years, it would likely not be impacted significantly by shrub increases in desert grassland. The foothills thornscrub-desert grassland ecotone may shift northward in Sonora, but the desert box turtle already lives in this tropical vegetation type near Arizpe.

With decreased summer precipitation and increased fall tropical storms, the hibernation-estivation inactivity period would be extended, breeding delayed until the monsoon rains started, and general conditions in this critical activity would be more stressful, reducing food resources, increasing mortality, etc.

Stresses: Increasing annual temperatures; decreased summer precipitation; prime grass-dominated desert grassland becomes more open as shrubs increase.

Responses: Hibernation-estivation inactivity begins later in fall, may or may not end earlier in summer; increase or decrease of summer precipitation will affect reproduction, food resources, survivorship, and dispersal/gene flow between metapopulations.

D-5.6 Ecological Status: Key Ecological Attributes and Indicators

This section of the conceptual model addresses Key Ecological Attributes and their potential indicators. The ecological status is a way of describing current status via criteria, functionality, or levels of attributes and asks if they are within the normal range of variation.

D-5.6.1 Key Ecological Attributes

Table D-6 identifies the key ecological attributes for this CE within the Madrean Archipelago ecoregion. A **key ecological attribute** of a focal ecological resource is a characteristic of the resource's biology, ecology, or physical environment that is critical to the resource's persistence in the face of both natural and human-caused disturbance. Alteration of such a characteristic beyond some critical range of variation will lead to the degradation or loss of the resource within decades or less. The KEAs table lists the identified key ecological attributes, with a brief definition, a rationale for why it is important for the CE, and a listing of stressors or change agents that might be affecting the key attribute.

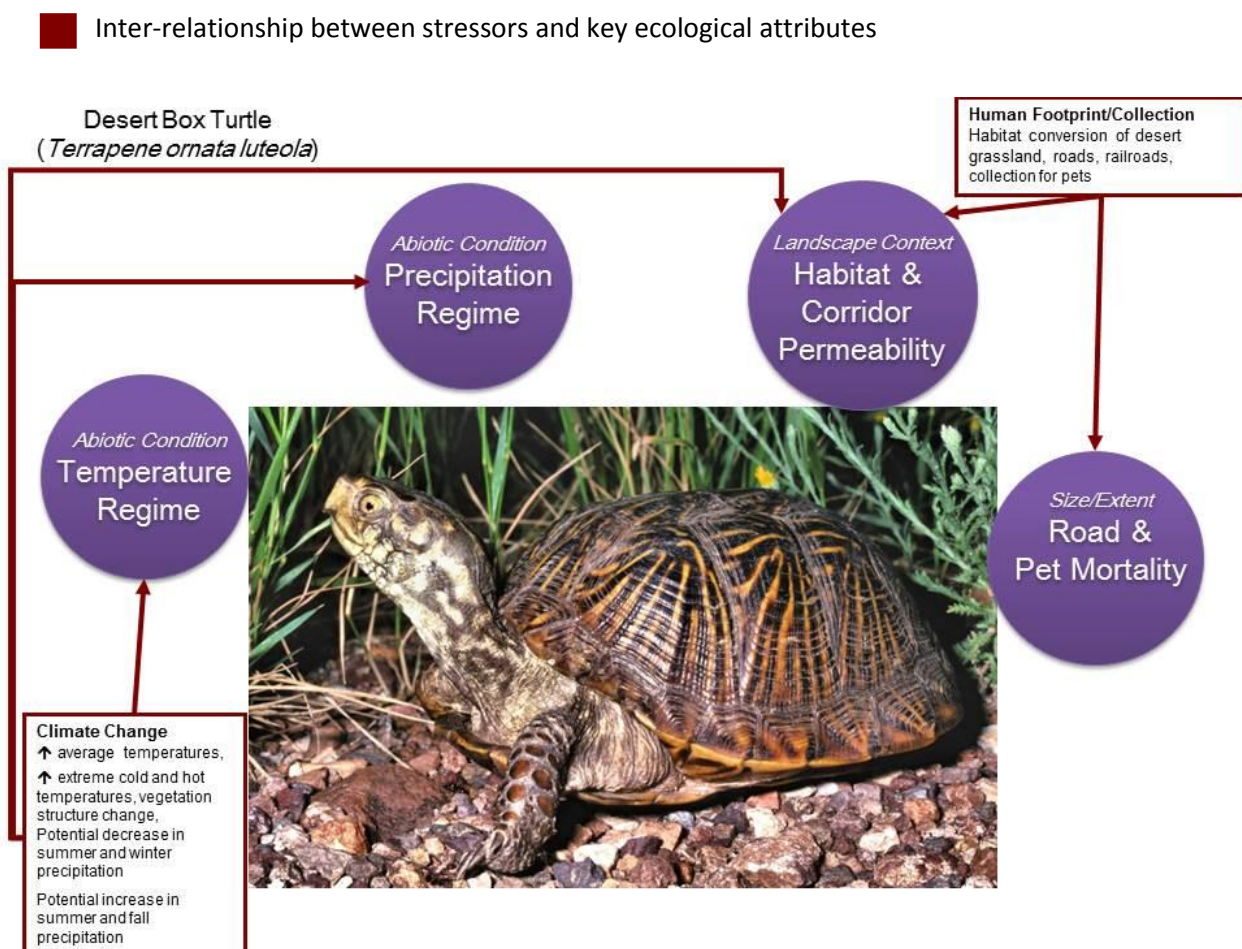
Table D-6. Key Ecological Attributes (KEA) related to the ecological status of the Desert Box Turtle (*Terrapene ornata luteola*) in the Madrean Archipelago ecoregion.

KEA Class: Name	Definition	Rationale	Stressors
Landscape Context: Habitat Availability and Quality	The extent of desert grassland habitat that is unfragmented by development and/or roads.	Ecological conditions and landscape dynamics are affected by land use. Land use impacts vary in their intensity where they occur, as well as their ecological effects with distance.	Factors that affect landscape condition in desert box turtle habitat can be abiotic (drought, fire) or related to human activities. Landscape level changes in desert grassland, such as large-scale urbanization and conversion of grassland to agricultural fields, have major impacts on desert box turtles populations. Grazing does not have a significant impact on the desert box turtle. Heavier grazing resulting in more open areas likely favors kangaroo rats and more preferred hibernation sites. Turtles may suffer direct mortality crossing roads.
Population Condition: Population Size & Demographics	The size and age classes of the turtle population.	Mortality and survivorship are opposing processes in population dynamics. Human related mortality stresses natural populations.	Animals killed on roads by vehicles; animals removed for pets. Development of roads fragmented desert box turtle habitats and gene pools. Changes in frequency and speed of vehicular traffic will also affect turtle mortality. Turtles are often seen on roads, especially during the monsoon season (Brennan and Holycross 2006). Many turtles are killed by automobiles, a continuous mortality on nearby populations. However, in most reptiles, males are far more likely to disperse away core population areas and cross roads.
Abiotic Condition: Precipitation regime	The mean annual and seasonal precipitation and fluctuations in them.	Turtles come out of burrows when the summer monsoons begin and turtle mating typically takes place during the monsoon season. Precipitation changes would also effect the availability of the turtle's food sources.	With decreased summer precipitation and increased fall tropical storms, the hibernation-estivation inactivity period for turtles would be extended, breeding delayed until the monsoon rains started, and general conditions in this critical activity would be more stressful, reducing food resources, increasing mortality, etc. This type of change may also affect the prime grass-dominated desert grassland if shrubs encroach. With increases both in summer precipitation and fall tropical storms, general conditions during the activity period from July to November would be less stressful better with increased food resources (insects, fruits, etc.), increased survivorship at all life stages, and potentially more successful dispersal and gene flow between metapopulations.
Abiotic Condition: Temperature regime	This mean annual and seasonal temperature and periodic extreme events.	Seasonal temperatures and periodic extreme events control the distributions of most animals and plants. Seasonal temperature changes dictate when turtles enter into hibernation in the fall.	With increased annual temperatures, desert box turtle would likely enter into hibernation later in the fall, shortening the hibernation-estivation inactivity period.

D-5.7 Conceptual Model Diagram

A conceptual model diagram for the CE provides a visual summary representation of the Key Ecological Attributes and species' life cycle (Figure D- 11). These diagrams are intended to show how various stressors interact with categories of Key Ecological Attributes to highlight important drivers for species ecological integrity. The arrows indicate relationships between stressors and but do not indicate the nature of the influence (i.e. positive or negative).

Figure D- 11. Conceptual model for the Desert Box Turtle, showing key ecological attributes (by class) for this species, and indicating relationships between stressors and KEAs.



D-5.8 References for the CE

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Amphibians

D-6 Chiricahua Leopard Frog (*Lithobates chiricahuensis*)

D-6.1 Classification Comments

Previously the North American leopard frogs (Figure D-12) were classified as *Rana pipens* with subspecies in major geographic areas. Later the subspecies were elevated to full species in the *Rana pipiens* Complex based on morphological and molecular analyses, mating calls, color patterns, and distributions. Additional species of leopard frogs were described subsequently, including *R. blairi* (Plains Leopard Frog) from the mid-United States west to New Mexico and Arizona (Mecham et al. 1973, Crawford et al. 2013, Rorabaugh 2008a). Frost and Bagnara (1976) described *Rana magnaocularis* (Northwest Mexico Leopard Frog) in western Mexico from Sinaloa to east-central Sonora. Platz and Mecham (1979) described *R. chiricahuensis* (Chiricahua Leopard Frog) from southeastern Arizona and adjacent Sonora. Platz and Frost (1984) described *R. yavapaiensis* (Lowland Leopard Frog) from central Arizona and northern Sonora. Platz (1993) described *R. subaquavocalis* (Ramsey Canyon Leopard Frog) from the Huachuca Mountains, Arizona, but is currently considered a synonym of *R. chiricahuensis* (Goldberg et al. 2004, Crother 2008). Frost et al. (2006) transferred most of the North American *Rana* to the genus *Lithobates*. Preliminary research revealed genetic differentiation between *L. chiricahuensis* along the Mogollon Rim in central Arizona and more southern populations (Platz and Grudzien 1999, Goldberg et al. 2004), suggesting the possibility of two species.

Species description - The Chiricahua Leopard Frog is recognized by the distinctive pattern on the rear of the thigh consisting of small, raised, cream-colored spots or tubercles on a dark background; dorsolateral folds that are interrupted and deflected medially; stocky body proportions; relatively rough skin on the back and sides; and often green coloration on the head and back. The species also has a distinctive call consisting of a relatively long snore of 1 to 2 seconds in duration. Snout-vent lengths of adults range from approximately 54 to 120 mm (2.1 to 4.7 in).

Figure D-12. Chiricahua leopard frog (*Lithobates chiricahuensis*) - Sycamore and Scotia Canyons, Arizona. Photo James C. Rorabaugh



D-6.2 Protection Status

The Chiricahua Leopard Frog was added to the list of category 2 candidate species for protection under the Endangered Species Act in November 1991 (56 FR 58804) and November 1994 (59 FR 58982; Clarkson et al. 1986, Clarkson and Rorabaugh 1989). Category 2 candidates were those taxa for which there was some evidence of vulnerability and threats, but lacked sufficient data to support a listing proposal. In 1994, the Chiricahua Leopard Frog was elevated to category 1 candidate status. Category 1 candidates were taxa with sufficient information on biological vulnerability and threats to support proposals to list them as endangered or threatened, but for which preparation of listing proposals was precluded by higher priority listing actions. In 1998, the Southwest Center for Biological Diversity submitted a petition to list this species as federally endangered, and followed with two lawsuits (SCBD 1998, 2013).

In 2002 the U.S. Fish and Wildlife Service listed the frog as Threatened (USFWS 2002). The species' recovery priority of 2C indicates a high degree of threat, a high potential for recovery, and a taxonomic classification as a species. A special rule exempted operation and maintenance of livestock tanks on non-Federal lands from the Section 9 take prohibitions of the Endangered Species Act (USFWS 2007). A recovery plan was completed in April 2007 (USFWS 2007). Critical habitat was designated in 2012 (77 FR 16324, March 20, 2012). Critical habitat includes a total of 10,346 acres (4,187 hectares) in Apache, Cochise, Gila, Graham, Greenlee, Pima, Santa Cruz, and Yavapai counties, Arizona, and Catron, Grant, Hidalgo, Sierra, and Socorro Counties, New Mexico. Safe Harbor agreements are in place throughout the range of the species in Arizona and southwestern New Mexico.

D-6.3 Biology and Distribution Status

D-6.3.1 Distribution

The species occurs in central and southeastern Arizona, west-central and southwestern New Mexico, and the Sky Islands mountain ranges and Sierra Madre Occidental of northeastern Sonora and western Chihuahua, Mexico. More specifically, it occurred historically in the mountains and valleys along the Mogollon Rim east of Camp Verde and the Verde River, but also in southeastern Arizona south of the Gila River from the Baboquivari Mountains east to the Peloncillo Mountains. Although still fairly well distributed through this range, the species has disappeared from about 88% of its historical localities in Arizona (Rorabaugh 2008b).

Systematic or intensive surveys for Chiricahua Leopard Frogs have not been conducted in Mexico (USFWS 2007). In northeastern Sonora, it is known in the Madrean Archipelago area from the Sierra San Luis-Sierra Pan Duro complex, the Sierra los Ajos, near Cananea (Sierra Elenita), and in the Río Santa Cruz south of Lochiel (the southern extension of the San Rafael Valley). A specimen from near Yécora in the Sierra Madre Occidental in east-central Sonora likely represents a western extension of the Chihuahua populations. The Chiricahua Leopard Frog is known in eastern Chihuahua from lowland areas from near Palomas on the New Mexico border south into the Sierra Madre Occidental to Durango (USFWS 2007, Rorabaugh pers. com, 2013).

D-6.3.2 Reproduction

Life cycle and reproduction - The life history of the Chiricahua leopard frog has a complex life cycle, consisting of eggs and larvae that are entirely aquatic and adults that are primarily aquatic (USFWS 2007). The species has a distinctive call and males can be temporarily territorial (USFWS 2007). Amplexus is axillary and the male fertilizes the eggs as the female attaches a spherical mass to submerged vegetation. Numbers of eggs in a mass range from 300 to 1,485 and may be correlated with

female body size (Jennings and Scott 1991, Sredl and Jennings 2005). Egg masses of Chiricahua leopard frogs have been reported in all months except November to January, but reports of oviposition in June are uncommon (Zweifel 1968, Frost and Bagnara 1977, Frost and Platz 1983, Scott and Jennings 1985). Eggs hatch in the wild after 8-14 days, depending on water temperature (USFWS 2007). Tadpoles are mainly herbivorous with faster growth rates in warmer conditions. Tadpoles have a long larval period, from three to nine months, and may over winter. Generally activity is between March and October (USFWS 2007).

Males reach sexual maturity at 2.1-2.2 in (5.3-5.6 cm) snout-vent length in less than a year (Sredl and Jennings 2005). Chiricahua leopard frogs presumably experience very high mortality (greater than 90 percent) in the egg and early tadpole stages, high mortality as newly transformed juveniles, and then relatively low mortality when the frogs are adults (Zug et al. 2001, USFWS 2007). Under ideal conditions, Chiricahua leopard frogs may live as long as 10 years in the wild (Platz et al. 1997).

D-6.4 Habitat and Ecology

D-6.4.1 Habitat

The Chiricahua Leopard Frog is an inhabitant of montane and river valley ciénegas, springs, pools, cattle tanks, lakes, reservoirs, streams, and rivers. The species requires permanent or semi-permanent pools for breeding and water characterized by low levels of contaminants and moderate pH (Watkins-Colwell and Watkins-Colwell 1998). Breeding sites require some open water and, ideally, vegetative cover with some bare substrate. Chiricahua Leopard Frogs can be eliminated from sites that become overgrown with cattails (*Typha* sp.) or other emergent plants. Frogs need some emergent or submerged vegetation, root masses, undercut banks, or fractured rock substrates as refugia from predators and extreme climatic conditions (Sredl and Jennings 2005).

Chiricahua Leopard Frogs also possess an unusual ability among members of the *Rana pipiens* Complex; they can darken their ventral skin under conditions of low reflectance and low temperature (Fernandez and Bagnara 1991, Fernandez and Bagnara 1993), a trait believed to enhance camouflage and escape predation (USFWS 2007). Uplands adjacent to water provide essential foraging and basking sites, particularly in riparian vegetation.

In Arizona, Sredl and Saylor (1998) found a significantly higher proportion (62 percent) of known extant populations in stock tanks as compared to those in riverine habitats (35%), suggesting Arizona populations of this species have fared better in stock tanks than in natural habitats. In some areas, stock tanks replaced natural springs and ciénegas or were developed at spring headwaters and now provide the only suitable habitat available to the Chiricahua leopard frog. Of the nine extant populations of Chiricahua leopard frogs on the Mogollon Rim in Arizona, one is a natural aquatic system and the others are artificial or highly modified aquatic systems (Sredl et al. 1997). The only known localities of the Chiricahua Leopard Frog in the San Rafael and San Bernardino valleys, Buckskin Hills, and the Patagonia Mountains of Arizona are stock tanks. In contrast, many stock tank populations in New Mexico were extirpated, apparently by disease (Painter 2000). Sredl and Saylor (1998) found that stock tanks in Arizona are occupied less frequently by non-native predators (with the exception of American Bullfrogs [*Lithobates catesbeianus*]) than natural sites. There is a high probability that the Chiricahua Leopard Frog would be extirpated from many more areas if ranchers had not built and maintained stock tanks for livestock production.

Although stock tanks provide refugia for frog populations and are important for this species in many areas, tanks only support small populations and are very dynamic habitats without habitat complexity. Tanks often dry out during drought, and flooding may destroy downstream impoundments or cause siltation, resulting in loss of aquatic communities and extirpation of frog populations. Construction of tanks may destroy natural habitats at or downstream of the tank, and may alter local hydrology. Periodic maintenance to remove silt from tanks may also cause a temporary loss of habitat and mortality of frogs. Most stock tanks do not provide suitable breeding habitat because they do not regularly hold water long enough for development of larvae to metamorphosis. Sredl and Saylor (1998) caution that stock tank populations are sometimes simply mortality sinks with little reproduction or recruitment.

Elevation: Chiricahua Leopard Frog localities range from 3,281 to 8,890 feet (ft) (1,000 to 2,710 meters [m]) elevation (Platz and Mecham 1979, Sredl et al. 1997, Smith and Chiszar 2003).

Vegetation: The Chiricahua Leopard Frog occurs in a wide variety of permanent and semi-permanent aquatic systems in pine-oak forest, oak woodland, chaparral, plains and desert grasslands, and even desert scrub habitats (Stebbins 1985). Deep areas, root masses, and undercut banks are used when escaping capture. Habitat heterogeneity is likely important, and vegetation adjacent to aquatic habitats is essential for foraging for prey and protection from terrestrial predators, mostly during the summer rainy season.

D-6.4.2 Landscape connectivity

A metapopulation is a system of local populations connected by dispersing individuals (Hanski and Gilpin 1991). Dispersal habitat provides routes for connectivity and gene flow among local populations within a metapopulation, which enhances the likelihood of metapopulation persistence and allows for recolonization of sites that are lost due to drought, disease, or other factors (Hanski and Gilpin 1991, USFWS 2007). Chiricahua Leopard Frogs will move rapidly into newly created suitable habitat rapidly near to occupied habitats (Sredl and Jennings 2005). The most likely dispersal routes include combinations of ephemeral, intermittent, and perennial drainages, as well as uplands. Some vegetation cover for protection from predators, and aquatic sites that can serve as buffers against desiccation (drying) and stopovers for foraging (feeding) are desirable along dispersal routes. A lack of barriers that would block dispersal is critical. Features on the landscape likely to serve as partial or complete barriers to dispersal include cliff faces and urban areas (USFWS 2007), reservoirs 20 acres (50 hectares) or more in size that are stocked with sport fishes or other nonnative predators, highways, major dams, walls, agricultural fields, or other structures that physically block movement (Todd and Andrews 2008, Eigenbrod et al. 2009). The effects of highways on frog dispersal can be mitigated with frog fencing and culverts (USFWS 2007). Although detailed studies of dispersal and metapopulation dynamics of Chiricahua leopard frogs have not been conducted, Jennings and Scott (1991) noted that maintenance of corridors used by dispersing juveniles and adults may be critical to conserving populations of frogs.

D-6.4.3 Food

Larval Chiricahua Leopard Frogs are primarily herbivorous. After metamorphosis, the frogs eat an array of invertebrates and small vertebrates (USFWS 2007). Available food items at one population site included bacteria, diatoms, phytoplankton, filamentous green algae, water milfoil (*Myriophyllum* sp.), duckweed (*Lemna minor*), and detritus (Marti and Fisher 1998). Adults eat invertebrates such as beetles, true bugs, and flies, but also fish and snails (Christman and Cummer 2006).

D-6.4.4 Phenology

Chiricahua Leopard Frogs are mostly active from March to October in the warm season with hibernation in the winter months and estivation in the arid foresummer (May-June; USFWS 2007). They can be

found active both day and night, but adults tend to be active more at night than juveniles (Sredl and Jennings 2005).

D-6.4.5 Competitors

Various other vertebrates potentially compete with the Chiricahua Leopard Frog for food and habitat. Historically, Chiricahua Leopard Frogs occurred together with native ranid frogs more frequently (James C. Rorabaugh, pers comm. 2013). Today Chiricahua Leopard Frogs occur with Lowland Leopard frogs at Peña Blanca Spring, Peña Blanca Canyon below Peña Blanca Lake, and Thumb Butte Tank. Chiricahua Leopard Frogs occurred with Plains Leopard Frogs in the Sulphur Springs Valley, but have not been found together in Arizona since the late 1980s. Both species were still together at Cuchillo Negro Warm Springs in Sierra County, New Mexico as recently as a couple of years ago.

Non-native animals such as American Bullfrogs, tiger salamanders, and centrarchid fishes are both predators (see above) and competitors of Chiricahua Leopard Frogs.

D-6.4.6 Demographics

The Chiricahua Leopard Frog is rare in suitable habitat, and local abundance appears to fluctuate greatly, and populations in stock tanks generally include fewer than 100 individuals (Santos-Barrera et al. 2004). Historically, it occurred at 212 sites in Arizona, 170 in New Mexico, and 12-13 in Mexico (USFWS 2007). It is now absent from many historical localities and numerous mountain ranges, valleys, and drainages within its former range (USFWS 2007). Where still present, populations often are few, small, and widely scattered (USFWS 2007). Some disappearances from historical sites probably represent natural fluctuations, but in most areas disappearances appear to reflect real, on-going declines caused by human impacts, (USFWS 2007).

Also of importance are degradation and loss of habitat as a result of water diversions and groundwater pumping, livestock management that degrades frog habitat, a history of fire suppression and grazing that has increased the likelihood of crown fires, mining, development, and environmental contamination; disruption of metapopulation dynamics; and increased chance of extirpation or extinction resulting from small numbers of populations and the dynamic nature of frog habitats (USFWS 2002).

D-6.4.7 Conservation Actions

Conservation efforts are being undertaken in Coconino, Tonto, Apache, Sitgreaves, Gila, and Coronado National Forests (Jennings 1995; Sredl et al. 1997). Both the northern and southern populations of *R. chiricahuensis* are listed as threatened under the Endangered Species Act in 2002, and a recovery team was established in 2003. Conservation actions include both short-term interim actions to prevent further deterioration of the species' status, and longer-term planning for eventual recovery of the species. Arizona Game and Fish Commission Order 41 prohibit the collection of this species from the wild in Arizona. It is included as Wildlife of Special Concern in Arizona (Arizona Game and Fish Department 1996). Priority research topics include identification of the importance of disease, pesticides and other contaminants, climate change, UV radiation, fire management, and possibly other threats to the status and recovery potential of the Chiricahua Leopard Frog. Also, research is needed on key aspects of the frog's status, distribution, and ecology.

D-6.5 Change Agent Characteristics

D-6.5.1 Non-native species predators

Numerous native vertebrate predators commonly coexist with the Chiricahua Leopard Frog, including fishes, the Sonoran mud turtle (*Kinosternon sonoriense*), other ranid frogs, and garter snakes (Rosen et al. 1996a, Platz and Mecham 1979, USFWS 2007). Tiger salamanders are native to the Chiricahua Leopard Frog's range in the San Rafael Valley in central Arizona (*Ambystoma tigrinum nebulosum*), southeastern Arizona (*A. t. stebbinsi*), and the mountains of Sonora, Chihuahua, and Durango (*A. rosaceum*). Native fishes, such as trout (*Oncorhynchus*), chub (*Gila*), longfin dace (*Agosia chrysogaster*), and topminnow (*Poeciliopsis*) occur within the range of the Chiricahua Leopard Frog.

Predation by non-native American Bullfrogs, fishes, crayfish (*Orconectes virilis* and possibly others), and tiger salamanders is implicated as a contributing factor in the decline of ranid frogs in western North America (Moyle 1973, Hayes and Jennings 1986, Bradford et al. 1993, Fernandez and Rosen 1996), and may be the most important factor identified so far in the current decline of the Chiricahua leopard frog (Rosen et al. 1994, 1996a). Barred Tiger Salamander (*Ambystoma tigrinum mavortium*) is introduced in southeastern Arizona and southwestern New Mexico.

In southeastern Arizona, Rosen et al. (1994, 1996a) documented 13 non-native predaceous vertebrate species in aquatic communities in the range of the Chiricahua Leopard Frog, including American Bullfrog, Barred Tiger Salamander (*Ambystoma tigrinum mavortium*), Largemouth Bass (*Micropterus salmoides*), trout, catfish and 8 other fish species. Rosen et al. (1994, 1996a) found that when American Bullfrogs and centrarchid fish (bass and Green Sunfish, *Lepomis cyanellus*) were present, Chiricahua Leopard Frogs were absent. Non-native vertebrate predators were absent from 16 of 19 localities where Chiricahua Leopard Frog localities. Chiricahua Leopard Frogs are rarely found with non-native predators in diverse and complex habitats with shallow water, vegetation cover, and other features that provide refuge from predators.

In the San Rafael Valley, Arizona, Chiricahua leopard frogs were only found at sites that lacked non-native fishes and American Bullfrogs (Snyder et al. 1996). In the White Mountains of Arizona, the disappearance of Chiricahua leopard frogs from most historical localities was correlated with the appearance of native tiger salamanders and non-native crayfish (Fernandez and Rosen 1996, Fernandez and Bagnara 1995). Crayfish were found to prey upon Chiricahua leopard frog larvae, metamorphs, eggs, and adults. Crayfish spread to the breeding pond of one of the last and possibly the most robust populations of Chiricahua leopard frogs in the White Mountains, Arizona (USFWS files, Phoenix, AZ; Fernandez and Rosen 1998), and have become very abundant in former Chiricahua Leopard Frogs habitats on the Blue River, Arizona (J. Rorabaugh, pers. comm. 2011).

Stresses: Introduction of non-native predators/competitors in habitats.

Responses: Decline or extirpation of population.

D-6.5.2 Disease

Chytridiomycosis is a fungal skin disease caused by the pathogen *Batrachochytrium dendrobatidis* (Bd) that is killing amphibians (mostly anurans) around the globe. Aquatic larval stages contract it, but it only attacks keratin. The only keratin in tadpoles is in the mouthparts, so larvae are not much affected (Berger et al 1998; Garmyn et al. 2012; James C. Rorabaugh, pers. comm., 2013). After metamorphosis, frogs and toads have keratin in their skin.

The extent of morbidity and mortality in amphibian species worldwide varies within and among populations due to factors such as ecological factors such as climate, elevation, host life history traits and host immunity due to genetic makeup and previous exposure (Savage and Zamudio 2011). Savage and Zamudio's study (2011) highlights the importance of genetic diversity to maintain populations in the face of this disease. Forrest and Schlaepfer (2011) found that lowland leopard frogs (*Rana yavapaiensis*) in Arizona that resided in water sources >30 degrees Celcius were significantly protected from infection by Bd. In locations where Bd was known to be present, infection rates in frogs dropped from 75-100 percent in water <15 degrees Celcius to less than 10 percent in water warmer than 30 degrees celcius. Temperature is also strongly linked to amphibian immune response (Forrest and Schlaepfer 2011).

Waterfowl, particularly wild geese, have been shown to be a carrier of Bd via their keratinous toe tissue and may play a role in dissemination of the disease (Garmyn et al. 2012). Water in which an infected frog is found should be considered contaminated for 7 weeks after removal of the infected individual (Johnson and Speare 2003).

Most die offs in frog populations in Arizona and Sonora occur during the winter months, late fall, or early spring, presumably when the frog's immune system is less able to fight off the Bd fungus. Such die offs called Postmetamorphic Death Syndrome (PDS) are characterized by death of all or nearly all metamorphosed frogs in a short period of time, leaving only tadpoles surviving in the population (Scott 1993). Dead or moribund frogs are often found during or immediately following winter dormancy or unusually cold periods. The apparent PDS of Tarahumara Frog (*Lithobates tarahumare*) populations were documented in southern Arizona and northern Sonora as early as 1974, and by 1983 this species had died out in Arizona (Hale et al. 2005).

Chytridiomycosis was documented in the Chiricahua Leopard Frog as early as 1992 (Santo-Barrera et al. 2004). Infected populations may exhibit periodic die-offs or be extirpated, but the Chiricahua Leopard Frog is persisting with the disease. In some areas that lack stressors (i.e., lower predation, higher population numbers, warmer waters, higher pH, and/or lower elevation), the disease appears to have little effect on population viability (USFWS 2007). Chytrid die offs have been seen in Chiricahua Leopard Frog populations in Catron County, New Mexico, as well as Arizona and Sonora. In some years, very few Chiricahua Leopard Frogs occurred in the canyons of the Santa Rita and Pajarito mountains in the spring, suggesting that frogs were dying during the winter months.

A number of other diseases may affect Chiricahua Leopard Frog although few have been documented in Chiricahua Leopard Frog populations. These include a ranaviruses confirmed present in a dead Chiricahua Leopard Frog, a bacteria that causes red leg and other pathogenic bacteria, fungal diseases such as Chytridiomycosis, Saprolegniasis, and Phycomycosis, and protozoans and metazoans among others (USFWS 2007).

Stresses: Increased chytridiomycosis fungal infections.

Responses: More frequent Postmetamorphic Death Syndrome die-offs of adults in the winter with loss of breeding for a year if population recovers (if all adults die overwinter, there is not spring breeding so tadpoles need to transform and breed – either the second or third year). Reduced probability of dispersal between populations.

D-6.5.3 Livestock management and habitat modification

Livestock grazing is the most widespread land management practice in western North America (Fleischner 1994). Intense livestock grazing during the late 1800's and early 1900's was likely a key cause of change in the structure and composition of montane forests, arroyo cutting and loss of ciénegas and riparian systems, increased shrub dominance in grasslands, and altered fire regimes (Hendrickson and Minckley 1984, Swetnam and Baisan 1996), although other factors such as groundwater pumping, logging, mining, loss of beaver populations, and climate change likely contributed (Hereford 1993, Bahre 1995a, b, Geraghty and Miller, Inc. 1995). Livestock are adapted to mesic habitats but are attracted to the high quality and quantity of forage and select riparian habitats for water, shade, and cooler temperatures, and can adversely affect riparian systems in a number of ways (Fleischner 1994, Belsky et al. 1999, Jones 2000).

As the human population has grown throughout the region and demand for water has intensified, aquatic ecosystems have been greatly altered (Kolar 2003). Severe fragmentation and alteration of aquatic habitats in the southwestern United States has likely constricted many wide ranging aquatic species into isolated pockets, and maintenance of aquatic corridors may be critical in preserving organisms in the arid Southwest (Jennings and Scott 1991). Numerous dams and intensive livestock grazing practices have changed water temperature and flow regimes, usually reducing habitat quality for native fishes (Rinne and Minckley 1991). Since the late 1800s and early 1900s, construction of earthen cattle tanks in upland drainages has been a common range management practice (U.S. General Accounting Office 1988). Livestock tanks provide water and aquatic habitats to many species of wildlife, including amphibians.

Livestock grazing is nearly ubiquitous within the historical range of the Chiricahua Leopard Frog, but grazing impacts on its populations are not well studied. The effects of livestock grazing on amphibian populations may be positive or negative (Jennings 1988, Rosen and Schwalbe 1998, Sredl and Saylor 1998). However, adverse effects to the Chiricahua Leopard Frog and its habitat may occur under certain circumstances as a result of livestock activities, including trampling of eggs, tadpoles, and frogs; deterioration of watersheds; erosion and/or siltation of stream courses; elimination of undercut banks that provide cover for frogs; loss of wetland and riparian vegetation and backwater pools; and spread of disease and non-native predators (Gunderson 1968, Arizona State University 1979, Hendrickson and Minckley 1984, Jennings 1988, Ohmart 1995, Jancovich et al. 1997, Belsky and Blumenthal 1997, Bartelt 1998, Belsky et al. 1999, Ross et al. 1999, U.S. Fish and Wildlife Service 2007, Sredl and Jennings 2005). Increased watershed erosion caused by grazing can accelerate sedimentation of deep pools used by frogs (Gunderson 1968). Sediment can alter primary productivity and fill interstitial spaces in streambed materials with fine particulates that impede water flow, reduce oxygen levels, and restrict waste removal (Chapman 1988), or fill pools entirely. In the dry season, low water levels in stock tanks and high detritus loads (including cattle feces), low water levels, high water temperature, and low concentrations of dissolved oxygen can result in high levels of hydrogen sulfide toxic to frogs and cause die-offs (Sredl et al. 1997).

Sredl and Howland (1994) speculated that distribution of extant Chiricahua Leopard Frog populations in Arizona might be reflective of habitat fragmentation and extinction without recolonization, as well as habitat quality.

Due to the cumulative effects of continued grazing by cattle and elk in central Arizona and west central New Mexico and other anthropomorphic stresses, riparian areas have been deemed the most damaged and threatened ecosystem in the Southwest (Fleischner 1994, Catron et al. 2000).

Other habitat modification taking place in the MAR includes spring development, road development, and channelization of previously pooled habitat.

Stresses: Degradation of habitats (increased soil erosion on slopes/siltation, silt removal in stock tanks, reduced water, etc.), degradation of upland habitat, conversion of natural habitats to stock tanks, contamination of water in habitats, increased habitat fragmentation.

Responses: Population declines or extirpation, less successful breeding in marginal habitats, increased predation by non-native species, reduced resistance to chytridiomycosis, reduced dispersal and gene flow between populations

D-6.5.4 Climate Change

Climate change is an ongoing process in the Southwest with potential effects on all species and biotic associations (Thompson et al. 1997). Mean annual temperatures rose 2.0-3.1°F in the American Southwest in the 20th century, and are predicted to rise 8.1-11.0°F in the 21st century (Southwest Regional Assessment Group 2000). Changes in precipitation are more speculative with contradicting predictions in different models (Southwest Regional Assessment Group 2000, Patterson 1997, Betancourt 2004). The effects of different climatic change scenarios in my view are discussed here.

Warmer mean annual temperatures – With annual warming there would be a general upward shift of biotic communities (desertscrub, grassland, oak woodland, pine-oak forest, and mixed-conifer forest). For montane woodlands and forests, there would be reductions in their areas as well. Mixed-conifer forests at the highest elevations would be most vulnerable. There could be a general conversion of plains grassland (San Rafael Valley, Animas Valley) to desert grassland. In desert grassland, shrub dominance should increase as perennial grass and herbaceous perennial dominance declines. Similar shrub increases occurred twice in the last 4,000 years during warm periods before European arrival (Van Devender 1995).

To the south in Sonora, desert grassland is replaced by foothills thornscrub where winter minimum temperatures increase. Tropical plants are very vulnerable to hard freezes. The northern distributional limits of many tropical species are in fingerlike northern tributaries of the Río Bavispe (in the greater Río Yaqui drainage basin) in northeastern Sonora, southeastern Arizona, and southwestern New Mexico. These species would move northward with warmer winter temperatures.

Hurricanes are formed where sea surface temperatures are 79.7°F (26°C) or higher (Data Discover Hurricane Science Center 2013). Today such storms can bring very large amounts of precipitation to Baja California, Sonora, and the Sky Island region in October and November. Storms are either part of the Bermuda high circulation that cross Panama and move northward through the Gulf of California or from the southern Pacific Ocean crossing Baja California to reach the mainland. With increased mean annual and sea surface water temperatures, tropical storms should reach the Madrean Archipelago more frequently with potentially large increases in precipitation.

Warmer temperatures, reduced winter precipitation, increased summer precipitation - Reductions in winter precipitation are expected as the North Pacific warms, the Aleutian Low cyclonic circulation pattern weakens, and fewer winter storm tracts cross California to the Sky Island Region. In the Madrean Archipelago region, reduction in winter-spring precipitation lengthens the arid foresummer in

May-June with related increases in forest fires and greater post-fire erosion. Shifts to greater summer precipitation dominance would have dramatic impacts on species and vegetation.

Winter precipitation gradually decreases southward in Sonora, and thornscrub and tropical deciduous forest (TDF) in southern Sonora are increasingly dominated by summer precipitation. However, winter drought influences successful spring reproduction of many tropical plants. In contrast, TDF from central Sinaloa southward to Costa Rica has very little or no winter precipitation. In the Sierra Madre Occidental and Sky Islands in northeastern Sonora, winter drought has serious impacts on 'summer evergreen Madrean' oaks (*Quercus chihuahuensis*, *Q. emoryi*, *Q. oblongifolia*, and others) and is an important abiotic limiting factor for the oak woodland-desert grassland ecotone.

Duration and amount of precipitation in the summer monsoon season increase southward into New World tropics, with more intense monsoon storms.

Warmer temperatures, reduced winter and summer precipitation - The same predictions for reduced spring precipitation discussed above apply in this scenario. Less summer rainfall would inhibit the northern expansions of tropical thornscrub species. Reductions of both winter and summer precipitation would increase impacts on desert grassland and montane woodlands and forests. Desert grassland would be even more prone to shrub increases with warm season shrubs like velvet mesquite (*Prosopis velutina*) favored over cool season subshrubs (*Gutierrezia*, *Isocoma*, other Asteraceae).

Greater climatic variability - In recent years, climate seems to be more variable with more frequent extreme events (Karl et al. 2009). Incursions of frigid Arctic air are particularly important in defining the transition between the New World tropics and northern temperate biomes at ca. 28°30' to 30°N latitude in east-central Sonora (Van Devender et al. 2010). Even with increasing mean annual temperatures and summer precipitation, freezes will have catastrophic impacts, including pruning or killing velvet mesquite, creosotebush (*Larrea divaricata*), succulents, etc. in desert grassland; pruning or killing tropical species in foothills thornscrub, limiting their northward dispersal; and dramatic impacts on plant phenology (flowering and seed production), reducing critical spring insect populations for animals.

Climate change is an ongoing process in the Southwest, but contradicting predictions about future precipitation regimes make evaluating threats to the Chiricahua Leopard Frog and developing recovery strategies difficult. Increasing temperatures have the potential to alter Chiricahua Leopard Frog breeding phenology, with potential direct effects of earlier reproduction in spring, more rapid development, shorter period of hibernation, longer period of estivation, changes in abilities to find food, spread of infectious disease, and changes in immune function (Blaustein et al. 2001, Beebee 2002). Increasing temperatures may affect the population dynamics of chytridiomycosis because the fungi's growth (Collins et al. 2003, Piotrowski et al. 2004) and effectiveness of antimicrobial peptides on the skin of ranid frogs (Longcore et al. 1999) are temperature dependent.

Increased summer precipitation would improve aquatic habitats and facilitate dispersal of Chiricahua Leopard Frogs between habitats. Yet, increased precipitation may provide more opportunities for predators to spread and adversely affect frog populations, offsetting any benefits due to more mesic conditions for Chiricahua Leopard Frogs.

If increased temperatures are coupled with reduced winter and summer precipitation, a variety of indirect effects could occur, including loss and fragmentation of Chiricahua Leopard Frog habitats. Winter precipitation is important in maintaining water levels in spring and early summer in montane

riparian habitats. Reduction in winter rainfall could be especially problematic for the Chiricahua Leopard Frog, which have been largely relegated to headwater canyons, springs, and stock tanks by the presence of non-natives in the larger aquatic systems. A lot of those smaller systems are at risk of drying out before the monsoons start, which heightens the importance of sites where water levels can be managed (James C. Rorabaugh, pers. com. 2013).

During drought, proximity of suitable drought-resistant habitats may be critical to persistence of each frog population. If Chiricahua Leopard Frogs cannot disperse from drying habitats and reach suitable habitat, droughts are likely to produce major population declines. Small drought refugia (crevices in concrete near an overflowing drinker, or accessibility to water storage tanks or drinkers) could become critically important for survival of frogs. Combined winter-summer drought would likely reduce habitat for and invasion by non-native predators.

Changes in interactions with prey, competitors, predators and parasites may be most serious adverse consequences of climate warming on amphibian populations. Drought could result in extirpations of Chiricahua Leopard Frogs as stock tanks and other marginal habitats dry up. Grazing impacts on habitats could be greater. Dispersal between populations would be more difficult, decreasing the recovery potential and survival of metapopulations.

Stresses: Increasing temperatures; reduced winter precipitation; increased summer precipitation; winter and summer drought.

Responses: earlier reproduction in spring, more rapid larval development, greater mortality of metamorphs and small adults, shorter hibernation period, extended arid foresummer with longer estivation period and increased forest fires and erosion; increased drying of essential headwater, spring, and stock tank habitats, declines and loss of populations; greater spread of chytridiomycosis fungal disease and non-native predators; limit dispersal and gene flow between populations.

D-6.6 Ecological Status: Key Ecological Attributes and Indicators

This section of the conceptual model addresses Key Ecological Attributes and their potential indicators. The ecological status is a way of describing current status via criteria, functionality, or levels of attributes and asks if they are within the normal range of variation.

D-6.6.1 Key Ecological Attributes

Table D-7 identifies the key ecological attributes for this CE within the Madrean Archipelago ecoregion. A **key ecological attribute** of a focal ecological resource is a characteristic of the resource's biology, ecology, or physical environment that is critical to the resource's persistence in the face of both natural and human-caused disturbance. Alteration of such a characteristic beyond some critical range of variation will lead to the degradation or loss of the resource within decades or less. The KEAs table lists the identified key ecological attributes, with a brief definition, a rationale for why it is important for the CE, and a listing of stressors or change agents that might be affecting the key attribute.

Table D-7. Key Ecological Attributes (KEA) related to the ecological status of the Chiricahua Leopard Frog (CLF) (*Lithobates chiricahuensis*) in the Madrean Archipelago ecoregion.

KEA Class: Name	Definition	Rationale	Stressors
Landscape Context: Habitat condition and complexity	This attribute is the intactness of permanent and semipermanent pools with associated upland habitat for foraging. It includes diversity of habitat structure within aquatic and upland areas.	The CLF requires permanent or semi-permanent pools for breeding with some open water and ideally some emergent or submerged vegetation. It also utilizes adjacent uplands for foraging and basking particularly in riparian vegetation.	Factors that affect landscape condition in CLF habitat can be abiotic (drought, fire) or related to human activities. Fire reduces cover in upland habitats and protective cover and food in riparian habitats, and increases erosion and siltation in CLF habitats. Modifications of hydrological systems in watersheds and groundwater pumping can affect the permanency of water in CLF habitats. Stock tank cleaning or abandonment can reduce or eliminate CLF populations.
Landscape Context: Intactness of suitable aquatic habitat	This attribute is the presence of aquatic habitat this is essential for successful breeding and population success. There are key differences in between stock tanks and other artificial waters that can provide refugia for frogs but that can only support small populations and natural waters with habitat complexity.	Grazing and related surface modifications affect general landscapes and riparian habitats in them. Water sources created or modified for grazing can create new aquatic habitat for the species or diminish the microhabitat quality at previously extant sites. Stock tanks can serve as refugia but can also be temporary habitat due to periodic maintenance, and may destroy natural habitat or dewater downstream habitat by altering local hydrology.	Grazing impacts the upland riparian habitat of CLF through reduced vegetative diversity, increased erosion related to reduced ground cover and siltation in aquatic habitats; and the reduction of cover plants adjacent to upland riparian habitat; and loss in food and increased predation. Some aquatic habitats are modified to hold water longer (benefiting CLF). Well-fed stock tanks create new CLF habitat and populations with different annual dynamics than natural populations
Landscape Context: Dispersal routes	This attribute is the presence of intact routes for dispersal that include combinations of ephemeral, intermittent, and perennial drainages as well as uplands and have some vegetation cover to protect against predators and aquatic sites for buffers against desiccation. They also require the absence of physical barriers to movement.	To maintain genetic integrity long-term, CLF populations are dependent on dispersal within metapopulations and from more distant populations. Dispersal corridors are important biological factors related to population genetic integrity, colonization, survival, disease and predation. They allow for gene flow between local populations within a metapopulation. Intact dispersal corridors allow for recolonization of sites when local populations are lost to disease, drought or other factors (Hanski and Gilpin 1991, USFWS 2007). Chiricahua leopard frogs are reasonably likely to disperse about one mile overland, three miles along intermittent drainages, and five miles along permanent drainages (USFWS 2007).	Disturbance of dispersal corridors stresses CLF populations. Features likely to serve as partial or complete barriers to dispersal include cliff faces, urban areas (USFWS 2007), reservoirs 20 acres (50 hectares) or more in size that are stocked with sport fishes or other non-native predators, highways, major dams, walls, agricultural fields, or other physical structures that block movement (Todd and Andrews 2008, Eigenbrod et al. 2009) Maintenance of corridors benefits CLF, but may also disperse of non-native predators, some of which are carriers of chytridiomycosis fungal.

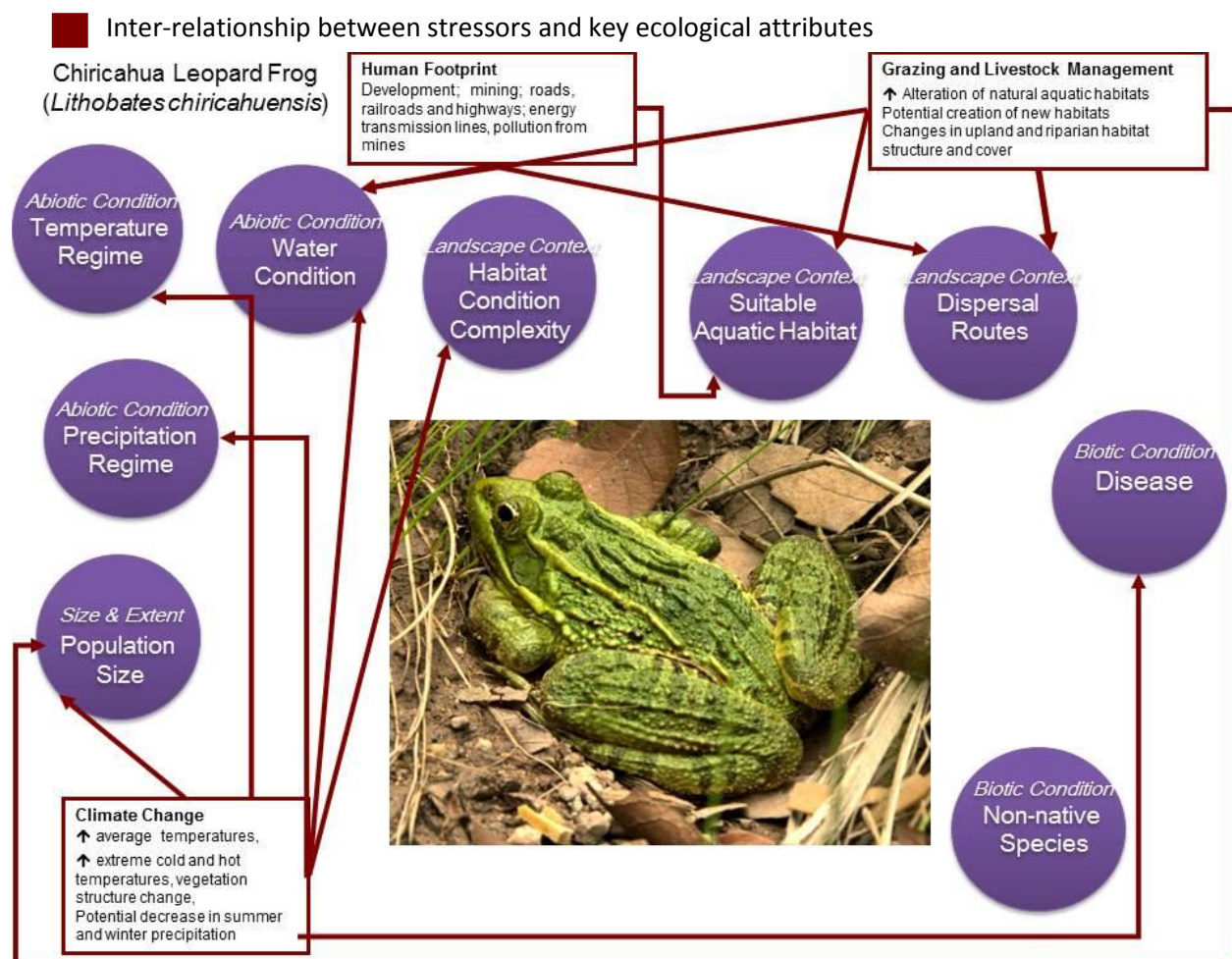
KEA Class: Name	Definition	Rationale	Stressors
Size/Extent: Population size & demographics	This attribute is the abundance of populations of CLF within a metapopulation, in addition to the abundance of individuals at a given breeding site.	Population size and demographic structure are direct indicators of the viability and health of the population, and can affect genetic diversity. Where still present, populations are often few, small and widely scattered (USFWS 2007). Of particular concern for CLF is the small numbers of populations within the landscape which increases the chance of extirpation or extinction.	Population size is influenced by various factors. Water quality and permanence, the presence of native and non-native predators (crayfish, fishes, bullfrogs, salamanders, etc.), and the presence of safe sites during drought affect the mortality of larval and adult CLF. In most areas, disappearances reflect real, on-going declines caused by human impacts (USFWS 2007). In stressed populations, CLF adults are more vulnerable to chytridiomycosis fungal infections. Population survival is more likely if there are good corridors connection sites within metapopulations.
Biotic Condition: Disease	This attribute is the abundance of chytridiomycosis fungus in the CLF population. In addition to measuring presence of chytrid in populations, the impact of the disease is increased by lack of adequate water and bank cover and presence of non-native predators that stress populations.	Disease can be an important factor in the biological condition of a species, affecting demography (mortality, survivorship). Infected CLF populations may exhibit periodic die-off or be extirpated.	A chytridiomycosis fungus (<i>Batrachochytrium dendrobatidis</i>) has been in CLF populations since at least 1992 (Santo-Barrera et al. 2004), but the species is persisting with the disease. The fungus attacks the skin of post-metamorphic frogs, and in stressed populations can cause severe mortality. Frogs are most heavily affected during or immediately following winter dormancy or unusually cold periods. Populations already stressed by predation, warmer waters, higher population density, lower water pH etc. may be more affected by the disease (USFWS 2007).
Biotic Condition: Non-native species	This attribute is the abundance of non-native predators that predate and compete with CLF in otherwise suitable CLF habitat.	Non-native species can be invasive with serious impacts on native species and biotic communities. Predation by non-native species is implicated as a contributing factor in the decline of ranid frogs in western North America (Moyle 1973, Hayes and Jennings 1986, Bradford et al. 1993, Fernandez and Rosen 1996), and may be the most important factor identified so far in the current decline of the Chiricahua Leopard Frog (Rosen et al. 1994, 1996a).	Non-native bullfrogs, fishes, crayfish and tiger salamanders predate and compete with CLF. Rosen et al. (1994, 1996a) documented 13 non-native predaceous vertebrate species in aquatic communities in the range of the Chiricahua Leopard Frog in southeastern Arizona, including American Bullfrog (<i>Lithobates catesbeianus</i>), Barred Tiger Salamander (<i>Ambystoma tigrinum mavortium</i>), Largemouth Bass (<i>Micropterus salmoides</i>), trout, catfish and 8 other fish species.

KEA Class: Name	Definition	Rationale	Stressors
Abiotic Conditions: Precipitation Regime	This attribute is the amount of precipitation received during the winter and summer seasons in addition to the mean annual precipitation.	The mean annual and seasonal precipitation and fluctuations in them are important abiotic factors in most animal and plant species. Precipitation is a very important in maintaining adequate aquatic habitats for the CLF. Breeding activity is generally between March and October (USFWS 2007). Winter precipitation is important in helping upland stream habitats persist during the arid foresummer in May-July. Summer monsoon precipitation is important because it supports maximum food insect production, and permits dispersal through otherwise dry corridors.	Decreased winter precipitation would lengthen the arid foresummer with related increases in forest fires and greater post-fire erosion. It could also be particularly problematic because CLF populations have been largely relegated to headwater canyons, springs, and stock tanks all of which are at risk of drying out before the monsoons start (J.C. Rorabaugh, pers. com. 2013). Decreased summer precipitation in the future would increase stress on CLF in most aspects of life history. Increased summer precipitation in the future could improve aquatic habitats and facilitate dispersal of CLF but could also facilitate dispersal of predators between habitats.
Abiotic Conditions: Condition of Water Sources	This attribute includes the temperature, chemistry, sediment and pollutant load in water in CLF habitat.	Water quality has a direct influence on aquatic animals in riparian habitats.	Pollutants in mountain streams from mining operations likely impact Chiricahua Leopard Frog populations. In the dry season, low water levels in stock tanks concentrations high detritus loads (including cattle feces), low water levels, high water temperature, and low concentrations of dissolved oxygen can result in high levels of hydrogen sulfide toxic to CLF and cause die-offs (Sredl et al. 1997).
Abiotic Conditions: Temperature Regime	This attribute is the mean annual temperature, seasonal average temperatures, and extreme highs and lows.	Seasonal temperatures and periodic extreme events control the distributions of most animals and plants. CLF activities including breeding, foraging, hibernation, and estivation periods are directly related to temperature (USFWS 2007).	Increased temperatures coupled with changes in precipitation could lead to loss and fragmentation of CLF habitats. Temperature changes may affect interactions with prey, competitors, predators and parasites which may have serious adverse consequences for CLF. The development of summer monsoonal precipitation is related to temperature. Increasing temperatures may affect the population dynamics of chytridiomycosis because the fungi's growth (Collins et al. 2003, Piotrowski et al. 2004) and effectiveness (Longcore et al. 1999) are temperature dependent. Decreased summer precipitation in the future would increase stress on CLF in most aspects of life history.

D-6.7 Conceptual Model Diagram

A conceptual model diagram for the CE provides a visual summary representation of the Key Ecological Attributes and stressors that affect the species' persistence (Figure D-13). These diagrams are intended to show how various stressors interact with categories of Key Ecological Attributes to highlight important drivers for species ecological integrity. The arrows indicate relationships between stressors and but do not indicate the nature of the influence (i.e. positive or negative).

Figure D-13. Conceptual model for the Chiricahua Leopard Frog, showing key ecological attributes (by class) for this species, and indicating relationships between stressors and KEAs.



D-6.8 References for the CE

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Assemblages

D-7 Grassland Birds

D-7.1 Introduction

This assemblage is meant to represent grassland birds at a level of detail that will help managers understand how key change agents may broadly affect grassland birds and to inform potential management responses. Vickery et al. 2009 defines grassland birds as "any species that has become adapted to and reliant on some variety of grassland habitat for part or all of its life cycle."

Understanding how grassland habitats contribute to the viability of grassland bird species, and how changes to that habitat may affect population health are critical to developing effective management strategies. Many grassland bird species are undergoing steep, widespread and long-term population declines and the grasslands they depend on are increasingly being lost and degraded through agricultural conversion, conversion to exurban and urban landcover, desertification and shrub encroachment (NRCS 1999, Pool et al. 2012). Some grassland bird species are neotropical migrants, but many are short distance migrants that winter primarily in the United States and Mexico.

Grassland ecosystems in the Madrean Ecoregion have been recognized for their regional biological value, especially for grassland birds (Biodiversity Support Program et al. 1995). Grassland birds have experienced a more consistent and geographically widespread decline in North America than any other guild of birds (Knopf 1994). In the Madrean Ecoregion there are a diversity of birds associated with grassland habitats that depend on this habitat for different aspects of their life histories, including overwintering, migration stopover, and breeding. Generally, species recognized as sensitive or of conservation interest to state and federal agencies managing resources in the Madrean Ecoregion are birds that breed in grassland habitat including Botteri's sparrow (*Peucaea botterii*), scaled quail (*Callipepla squamata*), western burrowing owl (*Athene cunicularia*), Aplomado falcon (*Falco femoralis*), northern harrier (*Circus cyaneus*), Arizona grasshopper sparrow (*Ammodramus savannarum*), loggerhead shrike (*Lanius ludovicianus*), and masked bobwhite (*Colinus virginianus ridgwayi*).

Grassland birds that overwinter in the Madrean Ecoregion include Baird's Sparrow (*Ammodramus bairdii*), ferruginous hawk (*Buteo regalis*), sage thrasher (*Oreoscoptes montanus*), and Sprague's pipit (*Anthus spragueii*). Botteri's sparrows are nearly endemic to the Chihuahuan Desert of northern Mexico and the adjacent United States and Baird's sparrows are concentrated in the region in the winter.

Species of conservation or management concern that are associated with healthy grasslands from the BLM Gila District (USDI-BLM 2010) include the Arizona Botteri's sparrow (*Peucaea botterii arizonae*), Arizona grasshopper sparrow (*Ammodramus savannarum ammoregus*), ferruginous hawk (*Buteo regalis*) (breeding population only), and western burrowing owl (*Athene cunicularia hypogaea*).

Additional grassland-associated birds in the region from lists compiled by Gori et al. (2012) include: Brewer's sparrow, Cassin's sparrow, chestnut-collared longspur, clay-colored sparrow, eastern meadowlark, western meadowlark, golden eagle, horned lark, lark bunting, lark sparrow, long-billed curlew, McCown's longspur, mountain plover, prairie falcon, sandhill crane, short-eared owl, and vesper sparrow.

Each grassland-associated bird has a unique set of habitat requirements and while there are some similarities, habitat management to benefit one specific species may not benefit other species. The

actual assemblage of grassland birds will vary with physical habitat structure, disturbances and other factors. Patterns of occurrence for individual species and large populations of wintering grassland-associated birds are highly dynamic from year to year probably due to the variability in rainfall patterns and food production, while birds that breed in the MAR stay mostly put. However, adequate cover of undisturbed grasslands is among the greatest factors affecting these populations (NRCS 1999).

Here we highlight four species selected as examples to characterize and illustrate some key patterns in the habitat requirements and life histories of these grassland-associated species (Arizona Botteri's sparrow (*Peucaea botterii arizonae*), scaled quail (*Callipepla squamata*), Arizona grasshopper sparrow (*Ammodramus savannarum ammoregus*), and Baird's sparrow (*Ammodramus bairdii*). In the desert southwest region, all of these birds depend primarily on the semi-arid grassland habitats of the Madrean Ecoregion. Additionally, we describe the major change agents for grassland birds, drawing upon examples from the species described here in some detail, as well as other grassland bird species in the region.

D-7.2 Individual Bird Highlights

NESTING SPECIES

D-7.2.1 Botteri's sparrow (*Peucaea botterii*)

Distribution. The range extends discontinuously from southeastern Arizona (Corman, in Corman and Wise-Gervais 2005), southwestern New Mexico, and extreme southern Texas southward to Nicaragua and Costa Rica (Howell and Webb 1995, Webb and Bock 2012).

One subspecies of Botteri's sparrow exists in the Madrean Ecoregion, Arizona Botteri's sparrow (*Peucaea botterii arizonae*). In southeastern Arizona, the species is locally common on the Sonoita Plains, eastern Santa Cruz and southwestern Cochise counties, between the Santa Rita and Huachuca Mountains; current range limits are Buenos Aires National Wildlife Refuge, Pima County, to the west; foothills of Mount Fagan south of Vail in Pima County, and Pinery Creek southeast of Dos Cabezas, Cochise County, to the north (not detected near Pinery Creek by Arizona Breeding Bird atlas surveys); and Hay Hollow Wash north of MacDonald Peak, 10 kilometers west of the entrance to Guadalupe Canyon, Cochise County, to the east (Phillips et al. 1964, Monson and Phillips 1981, Webb 1985, Webb and Bock 2012). Elevational range in Arizona is mainly 1,082-1,585 meters (Corman, in Corman and Wise-Gervais 2005). In extreme southwestern New Mexico, small numbers have been reported in recent years in May–August in Hidalgo County (Webb and Bock 2012).

This species is migratory in the United States and northern Mexico, where the species is present primarily from late April-May to September-October (Howell and Webb 1995). Wintering areas of these northern populations are little known (Webb and Bock 2012).

Habitat. These sparrows occur in semi-arid grasslands, particularly those with taller grasses, often in swales, floodplains, periodically flooded lower canyon bottoms dominated by giant sacaton (*Sporobolus wrightii*) (Webb and Bock 1990). Also important is proximity to grassy hillsides that do not have as tall of grasses (Webb and Bock 1990). They are most abundant in pockets of healthy ungrazed or lightly grazed grasslands (Webb and Bock 2012). Nesting occurs in dense, senescent stands of sacaton (often at upper edge of sacaton-dominated bottoms), and also in upland grasslands with very widely scattered low shrubs such as velvet mesquite (*Prosopis velutina*), whitethorn acacia (*Acacia constricta*), catclaw acacia (*Acacia greggii*), ocotillo (*Fouquieria splendens*), and low cacti, and sometimes they nest in rolling grasslands intermixed with open stands of evergreen oaks (Corman, in Corman and Wise-Gervais 2005).

In upland habitats, this species is associated with the thickest and tallest grasses, regardless of plant species (Bock and Bock, 1992). On the Audubon Research Ranch, south of Elgin, Arizona, Bock and Bock (1988) found a preference for monotypic stands of non-native Lehmann lovegrass, perhaps because this habitat is structurally more similar to sacaton habitat than are other upland grasslands dominated by native species.

Nests are on the ground and generally well concealed by a thick overhanging grass clump (Webb 1985). Foraging occurs mainly in open grassy sites where grasshoppers are more visible (Webb and Bock 1990). **Diet.** In summer, Botteri's sparrows eat primarily insects, especially Orthoptera, and also seeds (Webb and Bock 2012).

Reproduction. In Arizona, nesting peaks between late July and mid-August, with clutches recorded from mid-June to late August (Webb 1985), and nestlings sometimes present into late September (Webb and Bock 1996). Young leave the nest when approximately 10 days old, and adults feed fledglings for 4-5 weeks (Webb 1985).

Populations and Status. Prior to the 1900s, this species was more widely distributed in southeastern Arizona (Monson and Phillips 1981). Extended drought and human-caused habitat changes (see Change Agents) have reduced the species' distribution and abundance. Today, U.S. populations of this species are locally abundant, rebound from moderate habitat disturbances, and occupy a variety of grassland types; the species appears stable despite its restricted range (Webb and Bock 2012).

D-7.2.2 Grasshopper sparrow (*Ammodramus savannarum*)

Distribution. The breeding range extends from eastern Washington, southern British Columbia, southern Alberta, southern Saskatchewan, southern Manitoba, southern Ontario, southwestern Quebec, northern Vermont, New Hampshire, and southern Maine south to southern California, central Nevada, northern Utah, eastern Colorado, eastern New Mexico, northern Texas, Arkansas, northern Mississippi, Alabama, Georgia, North Carolina, and southeastern Virginia; and from southeastern Arizona, southwestern New Mexico, and southern Texas south to northern Sonora and northern Chihuahua; and also includes central Florida (Vickery 1996, AOU 1998). The main population is in the Great Plains, from North Dakota south to northern Texas, and east to Illinois (Johnson et al. 1998). The birds breeding in Arizona, New Mexico, Sonoran and Chihuahua populations are a separate subspecies (*Ammodramus savannarum ammoregus*).

The nonbreeding range extends from central California, southern Arizona, southern New Mexico, Texas, central Missouri, Tennessee, and North Carolina south through Mexico and Central America to northern Costa Rica, including the Bahamas and Cuba (Vickery 1996, AOU 1998).

This species is resident in Veracruz, Chiapas, Guatemala, Belize, Honduras, northwestern Costa Rica, Panama, Jamaica, Hispaniola, Puerto Rico, western Colombia, western Ecuador, and the Netherlands Antilles (Vickery 1996, AOU 1998).

In the Madrean ecoregion, a distinctive grasshopper sparrow subspecies (*Ammodramus savannarum ammoregus*) nests in southeastern Arizona (e.g., San Bernardino, Babocomari, San Rafael, Sulphur Springs, and Altar valleys, Sonoita Plains, extreme upper San Pedro River drainage), southwestern New Mexico (small population; Williams 2004), and adjacent northern Sonora and Chihuahua, primarily at elevations between 1,000 and 1,600 meters. The Sonoita and San Rafael valleys in Arizona and the Animas Valley in New Mexico are the primary population centers (Ruth 2008). Some of the Arizona subspecies remain in winter, whereas many likely move south into Sonora for the winter (Phillips et al.

1964), especially if grass cover and food resources are scarce (Corman, in Corman and Wise-Gervais 2005). In winter the other subspecies, *perpallidus*, that breeds in the Great Plains is commonly found in the Madrean Ecoregion.

Habitat. Breeding habitat generally consists of grasslands with large expanses of intermediate height grass, often with clumped vegetation interspersed with patches of bare ground, and with moderately deep litter and sparse coverage of low woody vegetation (Vickery 1996; Corman, in Corman and Wise-Gervais 2005). Nonbreeding habitats include grass-dominated fields, native prairie (Florida), and grazed pastures (Mexico and Belize) (Vickery 1996). In southeastern Arizona, Bock and Webb (1984) characterized the habitat as having nearly 75 percent ungrazed grass cover averaging 30 cm in height, 5 percent woody cover, and approximately 20 percent bare ground. Strong (1988) described the preferred habitat as having grass cover of 42-60 percent, shrub canopy cover of 1-8 percent, and an average grass height of 5 to 20 cm.

Nests are placed on the ground and usually are well hidden at the base of a grass clump, forb, or shrub (Vickery 1996).

These sparrows forage exclusively on the ground; exposed bare ground is critical for effective foraging (Vickery 1996).

Diet. Grasshopper sparrows eat mostly insects (especially grasshopper) in summer. In winter, the diet is primarily seeds (Vickery 1996).

Reproduction. In Arizona, most nesting occurs during the typical monsoon season. Adults carrying food to nestlings have been observed from late July to late August (Corman, in Corman and Wise-Gervais 2005).

Populations and Status. Breeding Bird Survey data (<https://www.pwrc.usgs.gov/bbs/>) through 2011 indicates a general declining trend in the western United States. BBS data for the Madrean ecoregion grasslands are too limited for reliable trend determination, but other survey data indicate that distribution and abundance likely have declined in recent decades (Ruth 2008). In New Mexico, numbers reported in 1992 in the Animas and Playas valleys were 109 and 41, respectively; in 2004, corresponding numbers in the Animas and Playas valleys were 21 and 0 (Williams 2004).

D-7.2.3 Scaled quail (*Callipepla squamata*)

Distribution. Scaled quail are resident (nonmigratory) from eastern Arizona, east-central Colorado, and southwestern Kansas south through western Oklahoma, western Texas, and interior Mexico to northeastern Jalisco, Guanajuato, Queretaro, Hidalgo and western Tamaulipas, with a large part of the range in northeastern Sonora, Chihuahua, Durango, and south to Hidalgo; introduced populations exist in central Washington and eastern Nevada (Howell and Webb 1995, AOU 1998, Dabbert et al. 2009).

Habitat. Habitat includes dry open grasslands with scattered low-growing shrubs such as burroweed, snakeweed, cacti, and mesquite, with plenty of open ground (Wise-Gervais, in Corman and Wise-Gervais 2005). In southern Arizona, 89 percent of sightings occurred in mesquite grassland, mixed shrubland, and shrub-dominated washes (Medina 1988).

In Arizona, these quails inhabit open plains, rolling hills, and low ridges and mesas, in open semi-arid grasslands consisting of perennial bunchgrasses scattered with low shrubs and cacti primarily at elevations between 853 and 2134 meters (Brown 1989). Occupied grasslands have a shrub component,

as long it is not too dense (Medina 1988, Guthery et al. 2001). King (1998) found that inhabited areas were dominated by perennial bunchgrasses, with 10% woody cover, whereas Bristow and Ockenfels (2006) reported use of areas with $\leq 10\%$ woody cover, $\geq 26\%$ grass cover, and relatively high grass species richness (birds were absent in areas containing $\geq 20\%$ woody cover). In southeastern Arizona, optimum habitat contains low-growing grasses, forbs, and shrubs with a ground cover between 10 and 50% (Goodwin and Hungerford 1977).

In New Mexico, this species prefers shrub-grass communities, using shrubland as available and avoiding grasslands devoid of shrubs; generally it is associated with mesa dropseed, a bunchgrass, three awn, honey mesquite, and broom snakeweed, avoiding sites dominated by stoloniferous black grama (Smith et al. 1996, Saiwana et al. 1998, Nelson et al. 1999, Joseph et al. 2003). In the Chihuahuan Desert in New Mexico, a mixture of mid- and late-seral conditions supported the greatest densities of quail (Smith et al. 1996, Nelson et al. 1997, Saiwana et al. 1998, Joseph et al. 2003).

Nests are on the ground, usually in protected, shaded sites in dense vegetation (Dabbert et al. 2009). In New Mexico and Arizona, quail nested under yucca (*Yucca elata*) and small bushes and occasionally in agricultural fields (Bendire 1892). In New Mexico, 66 percent of 14 nests were located in dead Russian-thistle, mixed forbs, soapweed, johnson grass (*Sorghum halepense*), and overhanging rocks (Russell 1932).

Diet. The diet includes seeds of forbs, shrubs, and grain; forb seeds and grains main fall and winter food, with insects and herbaceous leaves consumed on a seasonal basis (Dabbert et al. 2009).

Reproduction. Nesting generally begins in late April and May, but in dry years may be postponed until the summer monsoon begins in July or August (Brown 1989). Clutches averaging around 13 eggs are incubated for 22-23 days. Downy young have been seen as late as early October (Brown 1989).

Populations and Status. Scaled quail appear to be a “boom and bust” species that exhibits wide fluctuations in abundance (Dabbert et al. 2009). Breeding Bird Survey data (<https://www.pwrc.usgs.gov/bbs/>) through 2011 suggest that declines have occurred in the Arizona-New Mexico region since 1966 and during the most recent decade.

WINTERING SPECIES

D-7.2.4 Baird's sparrow (*Ammodramus bairdii*)

Distribution. The breeding range extends from southern Alberta, southern Saskatchewan, and southern Manitoba south to central and eastern Montana, North Dakota, northwestern and north-central South Dakota, and possibly western Minnesota, with unconfirmed records or possible nesting in Wyoming, Wisconsin, and extreme western Ontario (Green et al. 2002). The nonbreeding range extends from southeastern Arizona (Sonoita Plains, Altar and San Rafael valleys, base of Chiricahua, Huachuca, Santa Rita, and Patagonia mountains; Monson and Phillips 1981, Gordon 2000a), southern New Mexico (casual; Otero Mesa, Animas Valley; Hubbard 1978, Howell and Webb 1995, Green et al. 2002), and southwestern Texas south to northeastern Sonora, extreme northwestern Chihuahua, northeastern Durango, and extreme northern Zacatecas (Green et al. 2002).

Baird's Sparrows typically arrive in the Madrean ecoregion between September and mid-October and remain through early April.

Habitat. Nesting habitat (which is outside the Madrean ecoregion) includes ungrazed or lightly grazed mixed-grass prairie, prairie with scattered low bushes and matted vegetation (AOU 1998), local pockets of tallgrass prairie, wet meadows, and some types of disturbed habitats.

Nonbreeding habitat includes open grasslands and overgrown fields (AOU 1998). Specific nonbreeding habitat requirements are little known (Green et al. 2002). These sparrows are found mainly in dense stands of grass, usually in extensive grasslands, with a minor shrub component (Green et al. 2002). The birds seem to prefer areas of taller, denser grass, including grasslands dominated by several species of bunchgrasses (*Bouteloua* spp. and *Eragrostis* spp.) and few mesquite (*Prosopis velutina*) as only woody vegetation more than 1 meter in height (Gordon 2000a). They can also be found on south-facing slopes of mixed-oak grassland where the oaks are on the north slope. Baird's sparrows forage on the ground among grass clumps, usually well hidden from view (Green et al. 2002).

Diet. During the breeding season, the diet is dominated by various insects (Lane 1968). On the winter range in the Madrean ecoregion, the diet probably includes primarily the seeds of various grasses and forbs.

Reproduction. This species does not breed in the Madrean ecoregion. Most nesting occurs May-July (Green et al. 2002).

Populations and Status. In the breeding range, conversion of native prairie to cropland and exotic vegetation, invasion of native grasslands by exotic plant species, proliferation of shrubs due to fire suppression in moist portions of the range, and poor range management of some remaining tracts have greatly reduced Baird's sparrow populations from presettlement numbers (see Green et al. 2002). Breeding Bird Survey data (<https://www.pwrc.usgs.gov/bbs/>) indicate a significant decline since 1966 but an increase in abundance over the past decade (2001-2011). Christmas Bird Count data (<http://birds.audubon.org/christmas-bird-count>) in the winter range show low numbers of detections in Arizona and New Mexico, with no strong trend evident since 1966 or more recently.

D-7.2.5 Overview of Grassland Bird Habitats

The largest expanses of semi-arid grasslands occur in Arizona, New Mexico, and Texas, with the amount decreasing southward through Mexico. Most semi-arid grassland habitat is encompassed by privately owned cattle ranches. Some grassland habitats in the Madrean ecoregion are protected in reserves and experimental ranges, but these preserved areas represent a small percentage of the area dominated by semi-arid grasslands (Pool et al. 2012). For example, only about 8 percent of semi-desert grasslands in the U.S. portion of the Chihuahuan Desert is under some sort of protection (Pool et al. 2012).

Each grassland bird species has a unique set of habitat requirements (Table D-8). For example, Botteri's sparrows favor thick stands of tall grass whereas horned larks depend on expanses of short grass. Morrison (1999) found that in semi-desert grasslands of Arizona and New Mexico, woody plant cover strongly influences the presence and relative abundance of particular bird species. Changes in woody cover may cause recognizable shifts in bird assemblages at some level. Levels of woody cover <10% appeared to support sensitive grassland species such as grasshopper sparrows and chestnut-collared longspurs, and levels >10% support a different and more diverse set of species (Morrison 1999). Habitat conditions and grassland management favorable to one species may exclude other species.

The actual assemblage of grassland birds in a given area varies with physical habitat structure and surrounding landscape characteristics, such as:

- grassland vegetation density and growth form [short, medium, tall], and the presence or absence of woody vegetation (as described above) and other non-grassland features;
- the extent of unbroken grassland habitat; operative disturbance regimes;
- the presence of nearby water or wetlands;
- and surrounding land use.

Grassland landscapes with a diverse array of growth forms and successional stages support the highest diversity of grassland birds. Such diversity depends on natural processes that create a patchwork of different types of vegetation across a large landscape.

Table D-8. Examples of habitat characteristics for grassland bird species.

Species	Habitat Characteristics
Grasshopper sparrow (<i>Ammodramus</i> <i>savannarum</i>)	nearly 75% ungrazed grass cover averaging 30 cm in height, 5% woody cover, and approximately 20% bare ground (SE AZ; Bock and Webb 1984). grass cover of 42-60%, shrub cover of 1-8% percent, and average grass height of 5 to 20 cm (Strong 1988).
Botteri's sparrow (<i>Peucaea botterii</i>)	semi-arid grasslands, taller grasses, in swales, floodplains, periodically flooded lower canyon bottoms dominated by giant sacaton (Webb and Bock 1990). most abundant in pockets of healthy ungrazed or lightly grazed grasslands (Webb and Bock 2012). nesting: dense, senescent stands of sacaton often at upper-edge of sacaton-dominated bottoms), and upland grasslands with very widely scattered low shrubs; or in rolling grasslands intermixed with open stands of evergreen oaks (Corman, in Corman and Wise-Gervais 2005). in uplands is associated with the thickest and tallest grass, regardless of species (Bock and Bock 1992), including monotypic stands of non-native Lehmann lovegrass (Bock and Bock 1988) (structurally more similar to sacaton habitat than are other upland grasslands dominated by native species).
Scaled quail (<i>Callipepla squamata</i>)	perennial bunchgrasses with scattered low shrubs and cacti (Brown 1989). Typically occupied grasslands have a shrub component, as long it is not too dense (Medina 1988, Guthery et al. 2001). dominated by perennial bunchgrasses, with 10% woody cover (King 1998), ≤ 10% woody cover, ≥ 26% grass cover, and relatively high grass species richness (birds were absent in areas containing ≥ 20% woody cover) (Bristow and Ockenfels 2006) optimum habitat contains low-growing grasses, forbs, and shrubs with a ground cover between 10 and 50% (SE AZ; Goodwin and Hungerford 1977). shrub-grass communities, using shrubland as available and avoiding grasslands devoid of shrubs; avoiding sites dominated by stoloniferous black grama (NM; Smith et al. 1996, Saiwana et al. 1998, Nelson et al. 1999, Joseph et al. 2003).

Species	Habitat Characteristics
Baird's sparrow (<i>Ammodramus bairdii</i>)	<p>only winter [nonbreeding] in the Madrean Ecoregion; open grasslands and overgrown fields (AOU 1998).</p> <p>mainly in dense stands of grass, usually in extensive grasslands, with a minor shrub component (AZ; Green et al. 2002).</p> <p>seem to prefer areas of taller, denser grass, including grasslands dominated by several species of bunchgrasses with few mesquite as only woody vegetation more than 1 meter in height (Gordon 2000a).</p> <p>forage on the ground among grass clumps, usually well hidden from view (Green et al. 2002).</p>
Masked Bobwhite (<i>Colinus virginianus ridgwayi</i>)	<p>open savanna grassland located within river bottoms, level valleys and plains (Kuvlesky et. al. 1997).</p> <p>requires high humidity for breeding, typically delays nesting until the late summer monsoon period (Corman and Wise-Gervais 2005).</p>
Loggerhead Shrike (<i>Lanius ludovicianus</i>)	<p>tall woody plants are key for wintering areas, therefore shrub removal for grassland restoration may adversely affect this species unless groups of tall woody plants are retained (>2m height) which provide perching sites for hunting (Pool et al. 2012).</p>
Western Burrowing Owl (<i>Athene cunicularia hypogaea</i>)	<p>Associated with abundant burrowing mammals or other burrowing species of appropriate size (especially prairie dogs; also ground squirrels, badgers, marmots, skunks, armadillos, large species of kangaroo rats, or burrowing tortoises) (Haug et al. 1993)</p> <p>dry, open, short-grass, treeless plains, steppes, deserts, prairies (Haug et al. 1993)</p> <p>vegetation height within 50m of nest burrow best if <15cm</p> <p>quantitative vegetative characteristics of nest sites are variable locally, but e.g., % bare ground tends to be high, often >40%, and % grass and other vegetative cover tends to be low, often <40% for grass cover (see Green 1983, MacCracken et al. 1985, Green and Anthony 1989).</p>

This pattern is affected by annual and long-term climate variations. In the semi-arid grasslands of the Madrean ecoregion, the distribution and abundance of particular bird species are highly dynamic from year to year in accordance with variations in reproductive effort and success that in turn reflect variations in rainfall patterns and productivity of bird food resources (Pool et al. 2012). Long-term climate change exerts an effect through changes in habitat structure and function (e.g., disturbance regimes). It is important to note that grasslands that have low biodiversity may support unique species.

Overall, adequate cover of undisturbed grasslands for nesting and foraging is among the most influential factors affecting grassland bird populations (NRCS 1999). Generally, large blocks of undisturbed grassland can support breeding and feeding behaviors as well as provide winter and migration cover; to support an array of grassland-nesting birds species within an area, NRCS (1999) recommends the presence of contiguous grassland blocks of at least 500 acres in size.

Geographic variation in these habitat factors results in some specific areas in the ecoregion being richer in grassland biodiversity than other areas. For example, the Huachuca Mountains Grassland Valley Complex and the Sierra San Luis/Peloncillo Mountains, straddling the U.S. - Mexico borderlands, were ranked first and second among conservation areas for species richness and irreplaceability in the Apache Highlands Ecoregional Assessment (Marshall et. al. 2004).

D-7.3 Change Agent Characteristics

Change agents for grassland birds do not fit neatly into discrete categories. Instead, the factors that affect bird populations interact in various ways. For example, livestock grazing, fire exclusion, and drought often have interacted in exerting their effects on grassland birds. Livestock grazing and prescribed burning of grasslands often are integrated activities on some rangelands in the Madrean ecoregion. Also, invasion of non-native plant species may affect fire regimes. However, from a plant perspective, fire and grazing appear to affect species relatively independently (i.e., some species tend to be affected mainly by fire whereas others are more responsive to grazing) (Valone and Kelt 1999).

D-7.3.1 “Permanent” Habitat Loss and Fragmentation

Some habitat has been converted to intensive farming or urban/residential use and provides little or no habitat for grassland birds. For example, the favored sacaton habitat of the Botteri’s sparrow was reduced and fragmented during the twentieth century in part as a result of farming and human settlement (Webb and Bock 1990, 2012). In Arizona, some grasslands inhabited by grasshopper sparrow are in private ownership and rapidly being lost and fragment as they are developed into ranchettes, vineyards, and other suburban development (Corman, in Corman and Wise-Gervais 2005; Ruth 2008). Some of the former U.S. breeding range of the aplomado falcon (e.g., Animas Valley, New Mexico) has been detrimentally altered by conversion of grassland to farmland.

D-7.3.2 Shrub Encroachment

Declines of scaled quail distribution and abundance in grasslands in southeastern Arizona have been attributed to increases in shrubby vegetation or complete loss of native grass species (Rea 1973; Wise-Gervais, in Corman and Wise-Gervais 2005). Mesquite invasion in grasslands in southeastern Arizona has eliminated breeding populations of grasshopper sparrow in affected areas (Lloyd et al. 1998), and in the same region Cassin’s sparrows are more common in areas with reduced mesquite density (Maurer 1986). Morrison (1999) and Block and Morrison (2010) found that bird assemblages in semi-desert grasslands in southeastern Arizona and southwestern New Mexico changed greatly as woody plant cover increased; some grassland birds such as grasshopper sparrow (summer), chestnut-collared longspur (winter), and horned lark (both seasons) occurred in greatest numbers where woody plant cover (mainly mesquite and snakeweed) was less than 1 percent, whereas the richness of sparrow species during winter was highest among sites with 6-15 percent woody plant cover.

Some of the former U.S. breeding range of the aplomado falcon in the Madrean ecoregion has been detrimentally altered by an increase in mesquite. Habitat changes have directly impacted the falcon and some (e.g., degradation of grassland habitat by excessive cattle grazing) also have resulted in reductions in its avian prey populations. However, this falcon appears historically to have been a very rare or possibly irregular breeder in southern Arizona and New Mexico (Keddy-Hector 2000), so the degree to which habitat changes have affected it in this region are somewhat speculative.

Studies in Texas and New Mexico indicate that range improvements designed to promote grasses by brush control may remove important food sources for scaled quail (e.g., mesquite, snakeweed, broomweed; Ault and Stormer 1983, Davis et al. 1975, Leif and Smith 1993). An increase in perennial

grass cover in the Chihuahuan Desert region of New Mexico between 1961 and 1992 corresponded with a decline in the scaled quail population (Saiwana et al. 1998).

D-7.3.3 Habitat Alteration Associated with Livestock Grazing

In the semi-arid grasslands of the Madrean ecoregion, overgrazing and drought are thought to have caused a major reduction in grass cover that was accompanied by an increase in shrub density (especially of mesquite) and a reduction in fire frequency resulting from removal of fine fuels that carry fire (Chew 1982, Humphrey 1974, Pyne 1984, Brown and Archer 1989, Swetnam and Baisan 1996). Livestock grazing is recognized as a threat to bird species that are associated with semi-desert grassland in the Madrean ecoregion (Latta et al. 1999).

The causes of recent and long-term declines in scaled quail populations in the southwestern part of their distribution (Breeding Bird Survey data) are uncertain but are likely the result of habitat degradation, particularly overgrazing (Dabbert et al. 2009). In Arizona, Bock and Bock (1988) found more scaled quail on grazed sites compared with ungrazed sites, but overgrazing reduces residual grass cover, which is important to overwinter survival of scaled quail (Brown 1978). Grazing on early- and mid-seral condition rangelands appears not to be compatible with supporting scaled quail populations (Saiwana et al. 1998). Grazing during drought conditions may result in loss of cover needed to support quail populations (Saiwana et al. 1998, Joseph et al. 2003, Nelson et al. 1997).

The favored sacaton habitat of the Botteri's sparrow was reduced and fragmented during the twentieth century primarily as a result of overgrazing and trampling by cattle; water diversion and flood channelization associated with agricultural land use also contributed to the decline (Webb and Bock 1990, 2012). Botteri's sparrows tolerate moderate cattle grazing (Webb and Bock 1996).

Limited data suggest that Baird's sparrows have some tolerance of grazing. Gordon (2000b) found higher abundance in summer-grazed pasture than in a nearby area that had been idle for 30 years. However, this species elsewhere shows a preference for minimally grazed areas, and it may be absent from areas receiving more than a moderate amount of grazing (Green et al. 2002).

Grazing on semi-arid grasslands in Arizona tends to exclude grasshopper sparrows (Bock and Webb 1984). In southeastern Arizona, Cassin's sparrow and grasshopper sparrow are common in grasslands from which livestock grazing has been excluded for two decades, but uncommon on heavily grazed pastures (Bock and Bock 1988); Cassin's Sparrows are most abundant in ungrazed and lightly grazed grasslands with canopy coverage in excess of 50 percent (Bock and Webb 1984; Bock et al. 1986; Bock and Bock, 1988, 1992, 1999). Habitat degradation of southwestern grasslands, generally resulting from livestock grazing practices, has had a major negative impact on grasshopper sparrow populations (Saab et al. 1995, Ruth 2008).

D-7.3.4 Planting and Invasion of Non-native Forage Species

Roadsides are dispersal corridors for non-native grasses and other non-native plant species. These land uses and infrastructure features have facilitated the introduction and past and on-going spread of invasive species into the grasslands of the Madrean Ecoregion. In response to erosion and demand for cattle forage, species such as Lehmann's lovegrass (*Eragrostis lehmanniana*) and Boer lovegrass (*Eragrostis curvula*) were widely introduced into grasslands in southeastern Arizona and adjacent Sonora, MX, often at the expense of native bunch grasses. As a result, the spread of non-native perennial grasses within grasslands in this ecoregion has been substantial. As documented by Gori et al. (2012), Lehmann's lovegrass and, to a lesser extent, Boer lovegrass are common on at least 1.5 million

acres in this ecoregion; non-native grasslands with little to moderate woody increase now comprise 11% of the area's current and former grasslands. Other non-native grasses common in semi-desert grassland include fountaingrass (*Pennisetum setaceum*) and Natalgrass (*Melinis repens* ssp. *repens*), which have spread more slowly (Van Devender et al. 2007). Buffelgrass (*Pennisetum ciliare*) and soft feather pappusgrass (*Enneapogon cenchroides*) will probably invade further into desert grassland with warmer climates as native mesquites and succulents increase. Native grasses are still present in many invaded areas, but at a lower density.

These non-native forage and erosion control species can negatively affect grassland birds. For example, African lovegrasses (*Eragrostis* sp.) have been planted in an effort to restore defoliated ranges (Cable 1971, Freeman 1979). Scaled quail avoid stands of Lehmann's lovegrass, an exotic species planted extensively in Arizona and dominant at mid- to high elevations (Medina 1988). Compared to ungrazed native grasslands in southeastern Arizona, plantations of Lehmann's lovegrass are ornithologically impoverished, probably because they produce fewer seeds and insects (Bock and Bock 1988). In this study, total birds counted were about twice as many on native transects as compared to exotic transects. For example, Cassin's sparrows breed more abundantly in native Arizona grasslands than in grasslands dominated by exotic lovegrasses (*Eragrostis* spp.) (Bock and Webb 1984, Bock et al. 1986, Bock and Bock, 1988, 1992, 1999). Reduced abundance of birds in unburned grasslands dominated by exotic lovegrasses may be related to thick accumulation of dead grass litter, which may inhibit foraging (Bock and Bock 1992).

Expansion of native woody or subshrub species such as mesquite and snakeweed is also occurring in the Madrean Ecoregion; this change agent is discussed in the preceding section on shrub encroachment.

D-7.3.5 Rodent Control on Rangelands

Keystone species of mammals (primarily black-tailed prairie dogs in the Madrean Archipelago ecoregion) historically played a role in maintaining open habitats. Prairie dogs constantly clip vegetation to maintain an unobstructed view of the landscape and thereby help prevent the invasion of mesquite and other shrubs into semi-arid grasslands (Koford 1958, Van Auken 2000, Askins et al. 2007). Hence, the eradication of prairie dogs from southeastern Arizona and southwestern New Mexico through poisoning and habitat conversion by humans likely played a significant role in grassland conversion and loss, and no doubt it has affected a number of songbird species that depend on open grasslands. Additionally, burrowing owls—which do not actually burrow—are dependent on burrowing mammals for creation of suitable nest sites, and declines in burrowing owl populations in Madrean semi-arid grasslands certainly are at least in part a result of prairie dog extirpation. Extirpation of black-tailed prairie dogs from the region may have also been influential in eliminating an important prey species for the Aplomado falcon, its associated open hunting areas, and a dependent community of other raptors and corvids the nests of which are used by the falcons (J. Truett, pers. comm., cited by Keddy-Hector 2000).

D-7.3.6 Fire

In semi-arid grasslands, fire plays a mixed role for grassland birds, reducing cover in the short term and improving grassland conditions by reducing the density of woody species in the long run.

Bock and Bock (1988) found that fire had no effect on scaled quail numbers in a sacaton (*Sporobolus wrightii*) grassland in southeastern Arizona. Fall counts of scaled quail on burned and unburned grasslands were similar.

Botteri's sparrows tolerate prescribed burning (Webb and Bock 1996). Webb and Bock (2012) noted that most remnant sacaton stands are on private ranchlands, and that ranchers traditionally burn or mow sacaton fields annually in late winter (Feb) (to encourage new growth) and graze them in spring–summer (1 May–15 July) when upland grasses are still dormant (Cox and Morton 1986). Fire is suppressed at the margins of sacaton bottoms to prevent loss of upland grasses, leaving narrow strips of peripheral sacaton habitat intact. Because sparrow nests are concentrated at the edge of sacaton bottoms, only part of the colony is evicted during the first postfire breeding season; by second season, sufficient plant biomass is established to support moderate-sized colonies (Bock and Bock 1978, Webb and Bock 1990).

In Arizona, short-term declines in sparrow numbers may occur following prescribed burning in a mesquite-invaded semi-desert grassland (Kirkpatrick et al. 2002), but the long time-interval (~20 years) for grass fires benefit Botteri's Sparrow; birds may not reach maximum density in protected sacaton bottoms for <20 years after burning or other disturbance (Webb 1985). However, Webb and Bock (1990) noted that a sacaton stand that had not been burned or grazed for 30-50 years supported no Botteri's sparrows.

Grasshopper sparrows in Arizona avoided recently burned sites for two or more years postburn, and Cassin's sparrow were most abundant in unburned native grasslands with 50% canopy cover, avoiding burned areas for 2 years post fire (Bock and Webb 1984, Bock and Bock 1992).

Small breeding populations in New Mexico are threatened by shrub encroachment and spread of invasive species that may be a consequence of drought, heavy grazing, and a non-natural regime of fire and fire suppression (New Mexico Partners in Flight; <http://nmpartnersinflight.org/grasshoppersparrow.html>).

Long et al. (2012) found that nesting success of grassland birds in the southern Great Plains responded positively to prescribed burns of ungrazed shrub-encroached habitat during the dormant season, suggesting that fire exclusion has been a factor in shrub invasion followed by grassland bird declines.

D-7.3.7 Drought

Drought undoubtedly affects scaled quail populations at their range margins in southeastern Arizona (Wise-Gervais, in Corman and Wise-Gervais 2005). Population lows may be attributable primarily to widespread reproductive failures, possibly the result of inadequate rainfall and a lack of succulent foods (Bailey 1928, Wallmo and Uzzell 1958, Campbell et al. 1973, Brown 1989).

The primary factors posing threats to the future of grasshopper sparrows in the region appear to be loss and/or degradation of habitat due to exurban development, overgrazing, and the effects of long-term drought (Ruth 2008).

Based on their study in southeastern Arizona, Bock and Bock (1999) concluded that, due to loss of food and cover, the combined effects of drought and high-density, short-duration grazing left the land in a substantially denuded condition and had a strong negative effect on populations of ground-foraging, seed-eating birds wintering in the study area. Furthermore, one year of moderate rainfall and reduced stocking density only partially ameliorated these effects.

D-7.3.8 Disturbance (Nondestructive Intrusion)

Webb (1985) found that low-flying military aircraft, especially helicopters, and vehicular road traffic disturbed Botteri's sparrow females at nests; birds either flushed repeatedly or delayed returning for up

to 1 hour following disturbance. Attendant females exhibited intense alarm responses to bird blinds set up in nest vicinity by abandoning incubation for >1 hour and chipping repeatedly.

D-7.3.9 Overview of Change Agents

Grassland habitats in the Madrean Archipelago ecoregion have experienced significant habitat conversion or have lost the basic ecological processes (disturbance regimes) that sustained them. Fire suppression and the removal of native grazers (e.g., black-tailed prairie dog), appear to be particularly important. Many decades of grazing by domestic livestock have altered grassland condition, though light levels of grazing are compatible with grassland bird conservation. Some generalist bird species, and those tolerant of barren or shrub-encroached landscapes, persist in anthropogenic grasslands, but many grassland specialists have declined in the altered landscapes. Grassland birds are increasingly dependent on human management for habitat health. Together with adequate protection and favorable management of grasslands, annual and long-term variation in precipitation patterns plays a major role in the determining the distribution, abundance, and population trends of grassland birds.

D-7.4 Summary

Although the four species of grassland associated birds treated in detail (Botteri's sparrow, Arizona grasshopper sparrow, scaled quail and Baird's sparrow) have differing habitat requirements a common habitat need for all of them is relatively dense stands of native grass species with a limited woody cover component. The highest recorded tolerance of woody cover was recorded for scaled quail and it is noted birds were absent in areas containing $\geq 20\%$ woody cover (Bristow and Ockenfels 2006).

The intensity and distribution of cattle grazing particularly in combination with precipitation amount and timing leading to changes in grassland structure, diversity, density, and forage quality. Generally Botteri's, Baird's and Arizona grasshopper sparrows share a preference for dense stands of taller grasses. The current literature does not thoroughly define "dense" although for Grasshopper sparrow a lower level threshold of 42% grass cover was noted by strong.

Continuity of grassland habitats with different structural characteristics and with surrounding non-grassland habitats may also be an important focus for grassland-associated species as many species including the Botteri's sparrow and scaled quail rely on specific grassland habitat types for breeding cover while requiring other nearby habitat types for other components of their life history.

The diversity of habitat structure needs represented by this group of four grassland-associated bird species, points to the need to provide a diversity of grassland habitats distributed across the Madrean ecoregion. It is important to note that different combinations of focal grassland-associated species and management objectives will lead to different key components of grassland habitat and structure.

D-7.5 Ecological Status: Key Ecological Attributes and Indicators

This section of the conceptual model addresses Key Ecological Attributes and their potential indicators. The ecological status is a way of describing current status via criteria, functionality, or levels of attributes and asks if they are within the normal range of variation.

D-7.5.1 Key Ecological Attributes

Table D-9 identifies the key ecological attributes for this CE within the Madrean Archipelago ecoregion. A **key ecological attribute** of a focal ecological resource is a characteristic of the resource's biology, ecology, or physical environment that is critical to the resource's persistence in the face of both natural and human-caused disturbance. Alteration of such a characteristic beyond some critical range of

variation will lead to the degradation or loss of the resource within decades or less. The KEAs table lists the identified key ecological attributes, with a brief definition, a rationale for why it is important for the CE, and a listing of stressors or change agents that might be affecting the key attribute.

Table D-9. Key Ecological Attributes (KEA) related to the ecological status of the grassland bird assemblage in the Madrean Archipelago ecoregion.

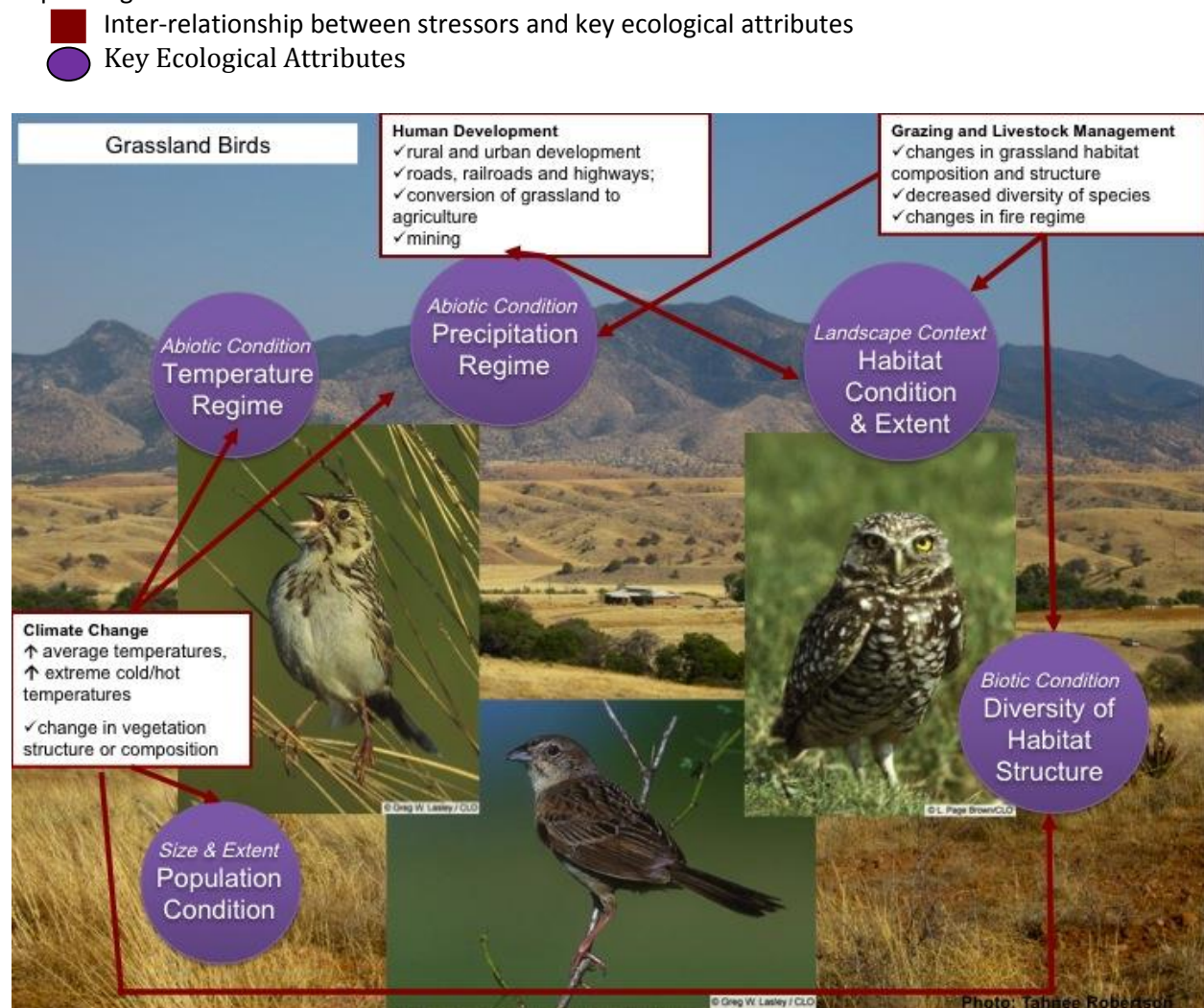
KEA Class: Name	Definition	Rationale	Stressors
Landscape Context: Habitat condition and extent	This attribute is the amount of unbroken blocks or contiguous patches of grassland habitat that is suitable for grassland bird overwintering and breeding.	Adequate cover of undisturbed grasslands is among the greatest factors affecting grassland bird populations (NRCS 1999). Grassland bird communities display an array of habitat needs. Of the diverse habitat needs of grassland birds examined in this model, the overall extent and intactness of all grassland habitat types is an important indicator for the assemblage.	Continued loss and conversion of grassland breeding and nesting habitats is the largest threat to most grassland birds (NRCS 1999). Factors that affect habitat condition and extent of habitat include human development in the form of urban and suburban development and agriculture, suppression of natural fire regimes, and livestock grazing and management. These stressors can destroy or alter suitable habitat.
Landscape Context: Diversity of habitat structure	This is the diversity of habitat features within a block of semi-desert grassland habitat including grass height, interspersions of non-grass habitat features such as woody shrubs or trees, cacti or bare ground.	Wintering grassland bird communities display high variability and composition from winter to winter and show low site fidelity in wintering grounds, highlighting the need to provide habitat distributed across the area of interest (Pool et al. 2012)	Stressors include conversion and loss of habitat due to development, agriculture and shrub encroachment, and reduction of habitat diversity due to livestock grazing.
Landscape Context: Intact Ecological Processes	The presence and intactness of ecological processes that drive grassland habitat structure including presence of native grazers and fire return interval.	The structure of grassland habitat including woody cover, grass species composition, grass height and other features is critical to supporting all aspects of life history for a variety of grassland-associated birds. These processes contribute to control of total woody cover and habitat renewal and diversity.	In desert grasslands in the Madrean Ecoregion native grazers including the black-tailed prairie dog are no longer present and fire regimes have been altered by human suppression and management and by changes in grassland fuel load due in part to grazing practices.
Size/ Extent: Population size & demographics	This attribute is the abundance of populations of grassland birds as well as the abundance of individuals at a given breeding site.	Population size and demographic structure are direct indicators of the viability and health of the population, and can affect genetic diversity.	Drought, pesticide use and competition with other species that are more tolerant of human caused changes are factors (Yosef 1996).

KEA Class: Name	Definition	Rationale	Stressors
Abiotic Condition: Precipitation Regime	This attribute is the amount of precipitation received during the winter and summer seasons in addition to the mean annual precipitation.	The mean annual and seasonal precipitation and fluctuations in them are important to grassland habitat productivity and, in combination with increasing temperatures may drive shrub/tree invasion of grasslands or grassland invasion of woodland habitat.	Decreased winter precipitation would lengthen the arid foresummer with related increases in fires and greater post-fire erosion. Changes in the arrival of monsoon season may adversely affect the phenology of a variety of grassland birds whose breeding is associated with the onset of monsoon and/or the wetness of winter. This may also drive changes in grassland composition that will alter what assemblage of species is supported in a given area. For example grasshopper sparrows and horned larks prefer habitat with <10% shrub cover (Morris 1999). Masked bobwhite are extremely sensitive to fluctuations in precipitation (Camou-Luders et al. 1999).
Abiotic Condition: Temperature Regime	This attribute is the mean annual temperature, seasonal average temperatures, and extreme highs and lows.		Increased temperatures coupled with changes in precipitation could lead to loss of and change in composition of grassland habitats; as well as temperature stress during breeding and nesting.

D-7.6 Conceptual Model Diagram

A conceptual model diagram for the grassland bird assemblage provides a visual summary representation of the Key Ecological Attributes and the species comprising the assemblage (Figure D-14).

Figure D-14. Conceptual model for the grassland bird assemblage, showing key ecological attributes (by class) for this assemblage of species, and indicating each KEA's influence, positive or negative depending on the KEA's condition.



D-7.7 References for the CE

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D-8 Nectivorous Bats: Lesser long-nosed bat (*Leptonycteris yerbabuenae*), Mexican long-nosed bat (*Leptonycteris nivalis*), Mexican long-tongued bat (*Choeronycteris mexicana*)

D-8.1 Taxonomic or Classification Comments

All three species are members of the Family Phyllostomidae, which comprises mostly tropical species. All three species have a long snout with a triangular-shaped flap of skin on the nose. They are about 2.5-3 inches long with a wingspan of about 13-17 inches. All have relatively large eyes and thus good vision. Long-nosed bats (*Leptonycteris*) lack an external, visible tail (Arita 2005; it is concealed and consists of just a few vertebrae); and *C. mexicana* has a very short tail (Ortega and Arita 2005).

Lesser long-nosed bat (*Leptonycteris yerbabuenae*): The nomenclatural history of this bat and the two other species in the genus has been rather confusing over the years. The currently accepted taxonomy for the lesser long-nosed bat is *Leptonycteris yerbabuenae* (Simmons and Wetterer 2002, Wilson and Reeder 2005, Cole and Wilson 2006). It is listed as Endangered under the U.S. Endangered Species Act (ESA) under the name *L. curasoae yerbabuenae* (Wilson and Reeder 2005).

Mexican long-nosed bat (*Leptonycteris nivalis*): Prior to the taxonomic revision by Davis and Carter (1962), specimens of *Leptonycteris yerbabuenae* were reported as *L. nivalis*.

Mexican long-tongued bat (*Choeronycteris mexicana*): This is the only species in the genus that comes as far north as the southwestern United States. Nominal subspecies *ponsi* of northwestern Venezuela is now recognized as a subspecies of *Choeroniscus periosus* (Wilson and Reeder 2005).

D-8.2 Individual Bat Highlights

D-8.2.1 Lesser Long-nosed Bat

Distribution and migration. *Global:* The global range extends from central California, southern Arizona, and southwestern New Mexico southward through the drier parts of Mexico (including Baja California) to Guatemala, Honduras, and El Salvador (Cole and Wilson 2006). Elevational range extends from sea level to 2400 meters (Arita 1993, 2005).

Significant maternity roost sites include Bluebird, Copper Mountain, and Old Mammon mines, Arizona; Bluebird, Copper Mountain, and Old Mammon mines, Arizona; Pinacate Cave, Pinacate Biosphere Reserve, Sonora; Cueva del Tigre, Carbo, Sonora; Sierra Kino—Isla Tiburon caves, Bahia Kino, Sonora; Santo Domingo Mine, Aduana, Sonora; Cueva La Capilla, cave near Buenavista, Baja California Sur; San Andres Island, Jalisco; Don Panchito Island, Jalisco; Cueva La Mina, Jalisco; Gruta Juxtlahuaca, Guerrero; Cueva Tempisque, Ocozocoautla, Chiapas; Tzinacanostoc, Puebla, Guerrero; and a cave near Tuxtla Gutierrez, Chiapas (Arita and Humphrey 1988; Ceballos et al. 1997; Cockrum 1991; Galindo et al. 2004; Horner et al. 1998; Stoner et al. 2003). Some of the nonmaternity roost sites include Patagonia Bat Cave, and Manila, Star of Texas, Blue Jay, Box Canyon, Buckelew, Catalina, Helena, Lone Star, Montezuma, Mustang, Papago Cave, Pyeatt, Tungstenand and Hilltop Mines, Arizona; a cave near Alamos, Sonora; Cueva Las Grutas, Michoaca'n; and Cueva de Xoxafi, Hidalgo (Arita and Humphrey 1988; Ceballos et al. 1997; Cockrum 1991). [Source: Cole and Wilson 2006]

Madrean Archipelago Ecoregion: The range in southern Arizona extends from the Picacho Mountains southwesterly to the Agua Dulce Mountains and southeasterly to the Galiuro and Chiricahua mountains, and southward into Mexico and beyond (Hoffmeister 1986). Also there are two late-summer records of immature individuals from the Phoenix area and one from the Pinaleno Mountains. A maternity colony formerly occupied Colossal Cave, southeast of Tucson; however, today all known maternity sites in Arizona are west of Tucson (Wolf, n.d.).

In New Mexico, the species is known from the Animas Mountains, Guadalupe Canyon, and Big Hatchet Mountains in Hidalgo County (Findley et al. 1975, Baltosser 1980, Cook 1986, Hoyt et al. 1994, Bogan 2005). One roost site in the Peloncillo Mountains on the Arizona-New Mexico border is known. Two roosts were documented in the mid-1990s in the Animas Mountains of Hidalgo County, New Mexico (Altenbach 1995). These include a new day roost that sometimes contains several hundred long-nosed bats, and an historical night roost where both species of long-nosed bat occur. One additional roost was found in the Big Hatchet Mountains in 2005 (Bogan 2005). Without specifying the species or location, Cryan (2012) reported the recent discovery of a significant new long-nosed bat roost on BLM lands, containing the largest aggregation of these bats found to date in New Mexico.

This species is migratory in the northern extent of its range and is not present in the United States during winter months. Migrants from central Mexico arrive in southwestern Arizona and adjoining portions of Sonora in April, form large maternity colonies. During the early part of their stay (late April to late July) pregnant females congregate at traditional roost sites, give birth, and raise their young at lower elevations within the range of columnar cacti. Males and perhaps nonpregnant females do not arrive until sometime in July. By late July, most females and young have dispersed from maternity colonies and some have moved to higher elevations where they are found feeding on agave flowers. Adult females and young of the year form late-summer roosts in southeastern Arizona. The bats return to central Mexico by September-October (Arizona Game and Fish Department 2011).

Lesser long-nosed bats follow two corridors in their northward spring migration (Coastal Lowland Route and Inland Montane Route). At least one corridor is utilized in the southward fall migration (Inland Agave Corridor), however it has not been confirmed that all bats using the Spring Coastal Lowland Route follow this Inland Agave Corridor for the return south in the fall (Wilkinson and Fleming, 1996).

Reproduction. Females arrive already pregnant at maternity roosts in Arizona as early as the second week in April. The single young is born in May [or June?]. Maternity colonies vary in size, from a few hundred to tens of thousands of females. Males maintain separate colonies during this period. Young bats can fly by the end of June, and the maternity colonies break up by the end of July (U.S. Fish and Wildlife Service 2013). This discrete, temporally concentrated birthing pattern exists in the Sonoran Desert from southern Sonora, Mexico, to Arizona. Farther south in Mexico, lesser long-nosed bats have a complex life history, with two female "demes," the reproduction of which is temporally and spatially displaced (Medellín 2005).

Habitat and Ecology. Habitats include tropical deciduous forest, as well as semi-deciduous thorn scrub, oak-pine, and cloud forests (Arita 2005). In Arizona, the species uses Palo Verde/Saguaro, Semi-desert Grassland, and Oak Woodland (Arizona Game and Fish Department 2011). The bats' diet of nectar enables it to be essentially independent of free water (Arizona Game and Fish Department 2011). Lesser long-nosed bats often roost deep in a mine or cave (Cryan and Bogan 2003).

Diet and Pollination. These bats feed primarily on nectar and pollen from the flowers of the saguaro cactus (*Carnegiea gigantea*) in early summer, then focus on the flowers of paniculate agaves from late summer to early fall. The bats may switch to less commonly consumed food plants or shift to distant locations when local food resources fail (USFWS 2007). Individual bats may land on a panicle of flowers to feed or they may bury their snout in a flower and rapidly lap up nectar while hovering (Arizona Game and Fish Department 2011). Feeding at agave flowers may often be done in groups. Long-nosed bats also consume the ripe fruits of saguaro and organ pipe cactus at the end of the flowering season (U.S. Fish and Wildlife Service 2013) and thereby serve as dispersal agents for the seeds. Insects may be ingested incidentally or intentionally while the bats feed on nectar. Lesser long-nosed bats also are known to feed on sugar water from hummingbird feeders at night; use of feeders may increase when natural food resources are scarce (USFWS 2007). Winter populations in southern Mexico feed on flowers of a variety of plants (Agavaceae, Bombacaceae, Cactaceae, Convolvulaceae, and Leguminosae) based on their temporal availability (see Cole and Wilson 2006).

After feeding, long-nosed bats go to night roosts, which may be different from day roosts, to rest and groom. As they groom themselves, they remove the pollen sticking to their fur with their claws and then lick it off their claws. This ingested pollen provides proteins and other nutrients not obtainable from nectar.

Lesser long-nosed bats pollinate various agave species, columnar cacti, and other Mexican plant species. Pollen collects on their heads and shoulders (sometimes making them look yellow) when they stick their head into a flower to get nectar. As they go from plant to plant, pollen is rubbed off on the pistils at each flower thus pollinating them. It is not yet clear just how important this bat is as a pollinator of saguaro and the agave species with which it is associated in the United States, since some populations of these plants also exist well outside the known range of this bat.

Activities, nightly movements and sociality. These bats emerge to feed about an hour after sunset. In Mexico, these bats fly up to 30 km each night from their roosts on Isla Tiburon in the Sea of Cortez to their feeding grounds in mainland Sonora (Arizona Game and Fish Department 2011). Long-nosed bats can travel 30 miles to forage, although probably most go no more than 15-20 miles. In southeastern Arizona, radio-transmitted bats were found to spend most of the night foraging, going to the same area each night. Between foraging bouts, they visited night roosts to digest and rest. Night roosts can be in man-made structures such as porches, barns, and bridges as well as caves, mines, and trees (Wolf, n.d.). Some observations indicate that lesser long-nosed bats spend about 6 hours a night foraging, alternating about 20 minutes of flying and feeding with about 20 minutes of roosting on plants or rocks and grooming.

Abundance and trends. This species is known to occur at only three maternity roosts in the United States, and in approximately 40 total roosts across its range in Arizona, New Mexico, and Mexico (USFWS 2007). Population size is not precisely known but appears to be at least several 10,000s in Arizona, in addition to those in Mexico (100,000s) and New Mexico (several thousand) (USFWS 2007). Based on improved and consistent monitoring protocols in recent years, the overall population at known roosts is believed to be stable or increasing in both the United States and Mexico (USFWS 2007). However, the number of known roost sites has not significantly increased (USFWS 2007).

Over the long term, populations are presumed to have declined significantly (Arizona Game and Fish Department 2011). However, the species is far more numerous than early reports (Wilson 1985) indicated (USFWS 1995). Cockrum and Petryszyn (1991) strongly disputed the reported decline of this

species and, in reviewing pertinent data, concluded that little evidence exists to document a long-term decline in Arizona, New Mexico, and Sonora; these authors stated "the various recent reports of disappearance appear to be, at least in part, the result of not looking in the right places at the right times" and further reported that "current populations...are little, if any, decreased from those of a quarter century ago. It even has been suggested that populations have increased in the past century because of more suitable roosts being available as a result of mining activity in the area."

In New Mexico, limited data indicate that lesser long-nosed bats can be locally numerous in late summer. In 1992, Hoyt et al. (1994) netted 150-200 *Leptonycteris* (primarily *L. yerbabuenae* but also about half as many *L. nivalis*), and believed that these represented a portion of a "much larger actual population of both species."

This species is listed as Vulnerable on the IUCN Red List, based on an assessment completed in 2008 (<http://www.iucnredlist.org/details/136659/0>). It is listed under the U.S. Endangered Species Act as as Endangered (under the name *Leptonycteris curasoae yerbabuenae*), but USFWS (2007) concluded that the species should be proposed for reclassification from endangered to threatened. *Leptonycteris yerbabuenae* was identified as a species of greatest conservation need in the Comprehensive Wildlife Conservation Strategy for New Mexico (NMDGF 2006) and the in State Wildlife Action Plan for Arizona (Arizona Game and Fish Department 2012).

D-8.2.2 Mexican Long-nosed Bat

Distribution and migration. *Global:* The range includes medium to high elevations in northern and central Mexico, southwestern Texas (southern Brewster and Presidio counties), and southwestern New Mexico (Frey 2004), at elevations of about 500 to 3,000+ meters. Most occurrences in Mexico are at elevations of 1,000-2,200 meters, but this bat been captured at an elevation of 3,780 meters (see Arita 1991), and the type specimen was caught near snow line at 17,816 feet (5,747 meters) on Mt. Orizaba, in Veracruz, Mexico (USFWS 1994). In Texas, the species is known from the Big Bend National Park and Chinati Mountain area. This species formerly was thought to occupy a much larger area, extending into southern Mexico and Guatemala, but specimens collected from those areas were assigned to *L. yerbabuenae* by Arita and Humphrey (1988). Simmons (in Wilson and Reeder 2005) described the range of *L. nivalis* as extending to southern Mexico and Guatemala, and she also included southeastern Arizona in the range. However, no actual records for Arizona are known, and Arita (1991) showed the range as extending only as far south as Puebla and northern Guerrero.

Madrean Archipelago Ecoregion: Two specimens of *Leptonycteris* taken in Hidalgo County, New Mexico (in 1963 and 1967), were determined to be *L. nivalis*. The presence of this species in New Mexico was reconfirmed with numerous captures of *L. nivalis* in the Animas Mountains in late August, Hidalgo County, in 1992; these bats may have been migrants from western Mexico (Hoyt et al. 1994).

The young are born in Mexico during April, May, and early June, then move northward with their mothers.

Reproduction. Adult females give birth to a single pup in spring (April-June) in Mexico before females arrive in the United States; no records of pregnant females are known from Texas (Schmidly 1977). In Texas, lactating females have been observed in June-July, flying juveniles in late June. Young are weaned in July or August.

Habitat and Ecology. Habitats in the northern part of the range in the United States and northern Mexico include desert scrub, open conifer-oak woodlands, and pine forests, generally in arid areas where agave plants are present (USFWS 1994). In Big Bend National Park, Texas, habitat includes mesquite-acacia (1800-4000 ft.), lechuguilla-creosotebush-cactus (1800-3500 ft.), deciduous woodland (3700-7800 ft.), pinyon-juniper-oak woodland (3700-7800 ft.), and cypress-pine-oak (5800-7200 ft.). In New Mexico, Mexican long-nosed bats inhabit upper desert scrub - pine oak woodlands in or near mountainous areas; characteristic vegetation in these areas includes agaves (*Agave* spp.), junipers (*Juniperus* spp.), oaks (*Quercus* spp.), and Mexican pinyon (*Pinus cembroides*) (NMDGF 1996). Specific roosting habitat requirements are not well known. Mexican long-nosed bats often roost near the entrances of caves and other roosts, in the “twilight zone” (Cryan and Bogan 2003).

Diet and Pollination. The diet includes mainly nectar and pollen of cacti (saguaro, organ pipe) and paniculate agaves; also insects associated with flowers, and probably some fruits, especially in the southern part of the range. When foraging at agaves, the bats crawl down the stalk, thrust the snout into the flowers, and obtain nectar with the long tongue (Harvey et al. 2011).

Activities, nightly movements and sociality. These bats emerge relatively late in the evening to feed (Harvey et al. 2011).

Abundance and trends. Long-term trend is unclear. Extent of occurrence and area of occupancy probably have not changed much, but the number of occurrences or subpopulations and population size may have dramatically decreased in some locations during the last three decades. Wilson (1985) found that this species was either completely absent or present in reduced numbers in known roosts. The number of bats found represented only a fraction of the total reported in previous studies. For example, in an abandoned mine in Nuevo Leon, Mexico, where an estimated population of 10,000 was observed in 1938, no individuals of *L. nivalis* were found in 1983 (Wilson 1985). Another mine in Nuevo Leon had a ceiling covered with newborn bats in 1967, but only one bat was found in 1983. A few other roosts had reduced numbers of bats compared to findings during previous surveys. These changes could indicate a decline in the overall population, but they might reflect movement of bats among different roosting sites in different years, or they could result from seasonal changes in bat distribution (survey dates varied).

Abundance at one of the two known roosting sites in the United States (a cave in Big Bend National Park) fluctuates widely from year to year (0 to 10,000+ individuals; Easterla 1972). In 2005, the greatest number of *L. nivalis* (at least 2742-2874, based on thermal imaging data) was present in July (Ammerman et al. 2009). Count data may not reliably reflect true numbers (e.g., some bats either roost in inaccessible crevices or are widely dispersed in the cave and thus cannot be observed and counted; England 2004), and no significant trend can be ascertained from available data. Reasons for the fluctuations are not understood, but they could reflect annual variations in food resources in different parts of the range (USFWS 1994). The estimated colony size of *Leptonycteris nivalis* at Emory Cave in July 2005 was similar in magnitude to numbers reported by Moreno-Valdez et al. (2004) for El Infierno Cave in Nuevo Leon, Mexico.

In New Mexico, limited data indicate that Mexican long-nosed bats can be locally numerous in late summer. In 1992, Hoyt et al. (1994) netted 150-200 *Leptonycteris* (primarily *L. yerbabuena* but also about half as many *L. nivalis*), and believed that these represented a portion of a “much larger actual population of both species.” Population trend in New Mexico is unknown (Hoyt et al. 1994).

This species is listed as Endangered on the IUCN Red List, based on an assessment completed in 2008 (<http://www.iucnredlist.org/details/11697/0>). It is listed under the U.S. Endangered Species Act as as Endangered. *Leptonycteris nivalis* was identified as a species of greatest conservation need in the Comprehensive Wildlife Conservation Strategy for New Mexico (NMDGF 2006)

D-8.2.3 Mexican Long-tongued Bat

Distribution and migration. *Global:* The range includes southern California, southern Arizona, southwestern New Mexico, the southern tip of Texas, and much of northern and central Mexico, including the Baja California peninsula, El Salvador, and Honduras in Central America (Arroyo-Cabral et al. 1987, Ortega and Arita 2005, Arizona Game and Fish Department 2006). Maternity colonies have been found in Arizona and New Mexico. A single individual has been found in Las Vegas, Nevada (Constantine 1987). In California, most records are from San Diego County in fall and winter, but occurrence records are relatively scarce (Pierson and Rainey, in Brylski et al. 1998). Elevational range extends from 300 to 3,600 meters (Arita 1993, Ortega and Arita 2005).

Madrean Archipelago Ecoregion: The range includes southeastern Arizona, from the Chiricahua Mountains to as far north as the Santa Catalina Mountains and as far west as the Baboquivari Mountains. Unpublished records (AZGFD) are available for Pinal, Pima, Graham, Santa Cruz, and Cochise counties (Arizona Game and Fish Department 2006).

This species is thought to migrate seasonally to take advantage of suitable sources of food. Populations in the United States normally migrate south across the border into Mexico to spend the winter. However, in the Tucson area, these bats have been observed at hummingbird feeders throughout the winter, so not all migrate to Mexico in the fall. Long-tongued bats have longer tongues than do long-nosed bats and may be able to exploit a wider variety of food plants (Wolf, n.d.).

Reproduction. Females segregate from the males. Only adult females move north from Mexico into the United States with the males remaining in the southern part of the range during the time that young are being nourished by the mothers in the north. Long-tongued bats form small maternity groups, usually of a dozen or less, but occasionally up to 50 adults. The young (typically one baby) are born mid to late June and early July, but parturition may be as late as September in Mexico (Harvey et al. 2011). As with many species of bats, the fetus is about 30% of the mother's weight. Young grow rapidly and can fly within probably 2-3 weeks. After the young become volant, these bats move about opportunistically in search of food. Females are known to carry their young in flight (Arizona Game and Fish Department 2006).

Habitat and Ecology. In the northern part of range in the United States, roost sites commonly are associated with mesic areas in oak-conifer woodlands or semi-desert grasslands. Dominant plant species include oaks (*Quercus*), alligator juniper (*Juniperus deppeana*), manzanita (*Arctostaphylos*), yucca, and agave. They also occur at higher elevations in pine-fir forests. A survey of historical roosts found that roosts occupied in 1999 were associated with surface water and riparian vegetation, and were near agaves, which are a major food source. Long-tongued bats commonly are found near roost-site entrances (Cryan and Bogan 2003); if disturbed, they tend to fly out into the open rather than deeper into the shelter (Wolf, n.d.). In Arizona and New Mexico, Mexican long-tongued bats exhibit high levels of roost-site fidelity (Mumford et al. 1964). They are often found in shallow caves and rock shelters and have been documented in the Cienega Creek Preserve in Arizona roosting in pocketed, eroded clay soil holes such as sink holes, or soil piping caves immediately adjacent to riparian habitat (AZGFD 2006).

Diet and Pollination. These bats feed on nectar, pollen, probably insects, and occasionally fruit of columnar cacti (Villa-R. 1967, Alvarez and Gonzalez-Q. 1970). Near Tucson, Arizona, long-tongued bats feed predominantly on cactus and *Agave* species (*Agave schottii* before mid-June, then *A. palmeri*) (Van de Water and Peachey 1997, Arizona Game and Fish Department 2006). The bristle-like tongue and lack of lower incisors aid this bat in lapping up flower nectar and pollen. During winter some are reported to feed at hummingbird feeders; it is not known if they are feeding on other things at that time (Arizona Game and Fish Department 2006). Cryan and Bogan (2003) observed species of *Agave* as the consistent floral characteristic of all sites visited, with *Agave schottii* observed blooming at occupied sites before mid-June (AZGFD 2006).

Long-nosed and long-tongued bats follow a "nectar corridor" of blossoming plants north from Mexico into the United States each spring (Fleming 2012). The spring migration of lesser long-nosed bats from central Mexico northward is thought to follow the sequential blooming of certain flowers from south to north (Arizona Game and Fish Department 2011). Mexican long-tongued bats apparently prefer to feed on *Agave* flowers during northward migration, and they arrive in the southwestern United States later than do the long-nosed bats (not until *Agave* has started blooming). At this time and into the fall both long-tongued and long-nosed bats feed primarily on *Agave*. During the winter in Mexico, *Choeronycteris* long-tongued bats apparently prefer the columnar cacti flowers, whereas *Leptonycteris* long-nosed bats prefer *Agave* (Arizona Game and Fish Department 2011) but feed also at the flowers of various other plants.

Activities, nightly movements and sociality. Activity peaks 1.5 hours after sunset and extends at low levels until about 3 hours after sunset. These bats are less gregarious than other colonial bats and less inclined to roost with other bat species. In roosts, they do not cluster closely together but hang 2.0-5.0 cm apart. Roosts usually consist of 15 or fewer individuals, but when considering roosts in close proximity to each other, population numbers may reach up to 40-50. These bats do not hibernate.

Abundance and trends. There is not sufficient evidence to conclude that populations in Arizona and New Mexico have increased or decreased in recent years. Surveys of 24 historically occupied sites in Arizona and New Mexico in 1999 found this species in 18 (75 percent) of the sites (Cryan and Bogan 2003). Sites where the species was not found were frequently disturbed, difficult to search, or historically occupied by single individuals. The authors concluded that populations of this species had not declined dramatically over the past several decades.

This species is listed as Near Threatened on the IUCN Red List, based on an assessment completed in 2008 (<http://www.iucnredlist.org/details/4776/0>).

D-8.3 Summary

Leptonycteris yerbabuena and *Choreonycteris* both rely on columnar cacti and agave species for nectar feeding but, in the Arizona portion of the Madrean ecoregion, *Choreonycteris* arrives later in Arizona when agaves are blooming and this is the primary food source for both *Leptonycteris* and *Choreonycteris* through the fall. This means that adequate agave foraging habitat in association with suitable daytime and nighttime roosts are important for both. The range of *Leptonycteris yerbabuena* and *Choreonycteris* overlap slightly, but *Choreonycteris* usually occupies higher elevations.

These bats spend most of the time in roosts, which fulfill many important roles in the bats' lives. The most important survival benefits of a suitable roost include: protection from the weather; protection from predators; cheaper thermoregulation; reduced commuting costs to foraging sites; improved

mating opportunities; improved maternal care; and competition avoidance, since few other vertebrates make use of most bat roost sites (Nolan, n.d.).

Long-term sustainability of food resources is extremely important to all three species and may be affected by habitat destruction/development, grazing/browsing, and fires (see Change Agents below). The degree of urbanization that a lesser long-nosed bat will tolerate to forage and move through is not well understood (AZGFD 2013).

It is important to note that there is a lack of comprehensive information on the extent of effects of agave harvesting, land clearing, grazing/browsing of agave and roost destruction or disturbance on the persistence and population health of nectar feeding bats. Because a large portion of the life histories of *Choreonycteris* and *Leptonycteris nivalis* are dependent on available roost sites and foraging habitat in Mexico including foraging habitat that supports seasonal migration, management actions taken within the Madrean ecoregion of the US may have limited effect on overall population numbers.

All three species are migratory and rely on corridors with adequate foraging habitat to support their seasonal movements. This is a critical part of their life histories, but because these species are at the northern edge of their range in the Madrean ecoregion, there is a limited extent of “nectar corridor” within the region.

D-8.4 Change Agent Characterization

This section of the conceptual model presents a narrative description of the primary change agents and current knowledge of their effects on nectar-feeding bat populations.

D-8.4.1 Roost disturbance and vandalism

Disturbance, vandalism, and outright killing of bats in roosting sites probably have detrimentally affected all three nectar-feeding species (USFWS 1994). Excessive disturbance may lead to increased mortality (e.g., of neonates) and roost abandonment. Activities associated with attempts to control vampire bats in Mexico have resulted in direct mortality and roost abandonment by long-nosed and long-tongued bats, and suitable alternate roost sites for displaced bats may not be available, even if other caves or abandoned mines are present. These bats have low reproductive rates and can recover only slowly from population declines.

Disturbance impacts and damage to roost sites from human-related activities such as recreation and caving are probably not as great as previously thought, but the potential for such impacts will likely increase in the near future due to increasing development, urbanization, and other land-conversion activities, all of which result in additional human presence in the areas where long-nosed bat roost sites are located (USFWS 2007).

D-8.4.2 Closure of cave and mine entrances

Intentional closing of caves and mines due to human safety concerns has eliminated some roost sites and could eliminate additional sites in the future. Natural cave-ins or gradual fill-in of roost-site entrances or passages also may close off bat access or render some entrances or important passages too small for a large bat colony to pass through without greatly increasing the danger of predation; however, these natural processes are not regarded as a major threat to any of the nectar-feeding bats.

D-8.4.3 Installation of improper gates

Erection of gates or other semipermeable structures to exclude humans may unintentionally exclude bats from their roosts or degrade or eliminate roost habitats due to changes in air flow and modification

of the climate inside the caves or mines. Additionally, modifications that affect bat movements inside or outside caves or mines may result in increased predation on bats (Nolan, n.d.). The response of long-nosed bats to various gate designs needs further research (USFWS 2007). State of Texas Mine has been gated and research conducted on the response of Lesser long-nosed bats to the gate.

D-8.4.4 Loss of forage plants

Destruction of forage plants through excessive human harvesting of native agaves for food and alcohol (e.g., tequila) production, particularly in Mexico, is a threat to these bats (USFWS 1994). Legitimate tequila producers likely have minimal effects on natural forage availability, but bootleg production often utilizes wild agaves and remains a threat (USFWS 2007). Clearing of land, grazing/browsing of agave species, and fires that remove foraging habitat in Mexico may also threaten these species. Recent evidence indicates that lesser long-nosed bats can change their foraging patterns and food sources in response to periodic loss or great reduction in a usually important plant species; this species appears to be more resourceful and resilient than previously thought (USFWS 2007).

Browsing on newly emergent flower stalks of agave species has been suggested as decreasing foraging opportunities (AZGFD 2006). For lower elevation species (*Leptonycteris yerbabuena*) relying on foraging patches in desert and grassland habitats, habitat loss due to development or land use conversion may pose an important threat. For species typically found at higher elevations (*Leptonycteris nivalis*) grazing/browsing of forage plants and loss of forage plants due to fire may pose an important threat. The level of tolerance *Leptonycteris yerbabuena* has for foraging in suburban, or urban habitat patches is not currently known (AZGFD 2013).

D-8.4.5 Illegal border activities

These activities, and associated enforcement actions along the U.S.-Mexico border, affect roosts through disturbance and destruction, and they affect foraging habitat through vegetation damage and increased potential for fire (USFWS 2007). In part as a result of border activities, threats to bat roost sites continue and, in fact, have likely increased in recent years (USFWS 2007).

D-8.4.6 Wind energy

A wind farm project has been proposed on Fort Huachuca, Arizona, an area supporting at least five late-summer lesser long-nosed bat roosts and agave foraging areas. No current wind farms occur in areas occupied by nectar-feeding bats, so the impacts on these bats are uncertain (USFWS 2007). Mortalities at wind turbines have been well documented for other bats species.

D-8.4.7 Invasion of non-native plant species

Expansion of non-native plant species affects the availability of bat food resources over most of their ranges (USFWS 2007), but little specific information on this appears to be available.

D-8.4.8 Increased fire frequency and intensity; drought

Fires in recent years on Cabeza Prieta NWR, and in the Santa Rita, Catalina, Huachuca, and Chiricahua mountains have impacted large areas of lesser long-nosed bat foraging habitat. Increased human presence in the range of the nectar-feeding bats, and continued drought in the region, in addition to the presence of non-native plant species, may result in more frequent and intense fires and could lead to further negative impacts of bat foraging areas.

Ongoing, long-term drought may result in increased failures in the bats' saguaro and agave food resources (USFWS 2007). On at least a short-term (single season) basis, lesser long-nosed bats appear to

be able to shift to other plant species (USFWS 2007), but the long-term effects of continued drought are uncertain.

D-8.4.9 Information Needs

For all 3 species, better information is needed on current distribution and abundance, trends, migration patterns, interactions with and dependence on specific plant species, and characteristics of preferred roost sites. Additionally, better information is needed on gate design and its effects on nectar-feeding bats although State of Texas Mine has been gated and research conducted on the response of Lesser long-nosed bats to the gate.

D-8.5 Ecological Status: Key Ecological Attributes and Indicators

This section of the conceptual model addresses Key Ecological Attributes and their potential indicators. The ecological status is a way of describing current status via criteria, functionality, or levels of attributes and asks if they are within the normal range of variation.

D-8.5.1 Key Ecological Attributes

Table D-10 identifies the key ecological attributes for this CE within the Madrean Archipelago ecoregion. A **key ecological attribute** of a focal ecological resource is a characteristic of the resource's biology, ecology, or physical environment that is critical to the resource's persistence in the face of both natural and human-caused disturbance. Alteration of such a characteristic beyond some critical range of variation will lead to the degradation or loss of the resource within decades or less. The KEAs table lists the identified key ecological attributes, with a brief definition, a rationale for why it is important for the CE, and a listing of stressors or change agents that might be affecting the key attribute.

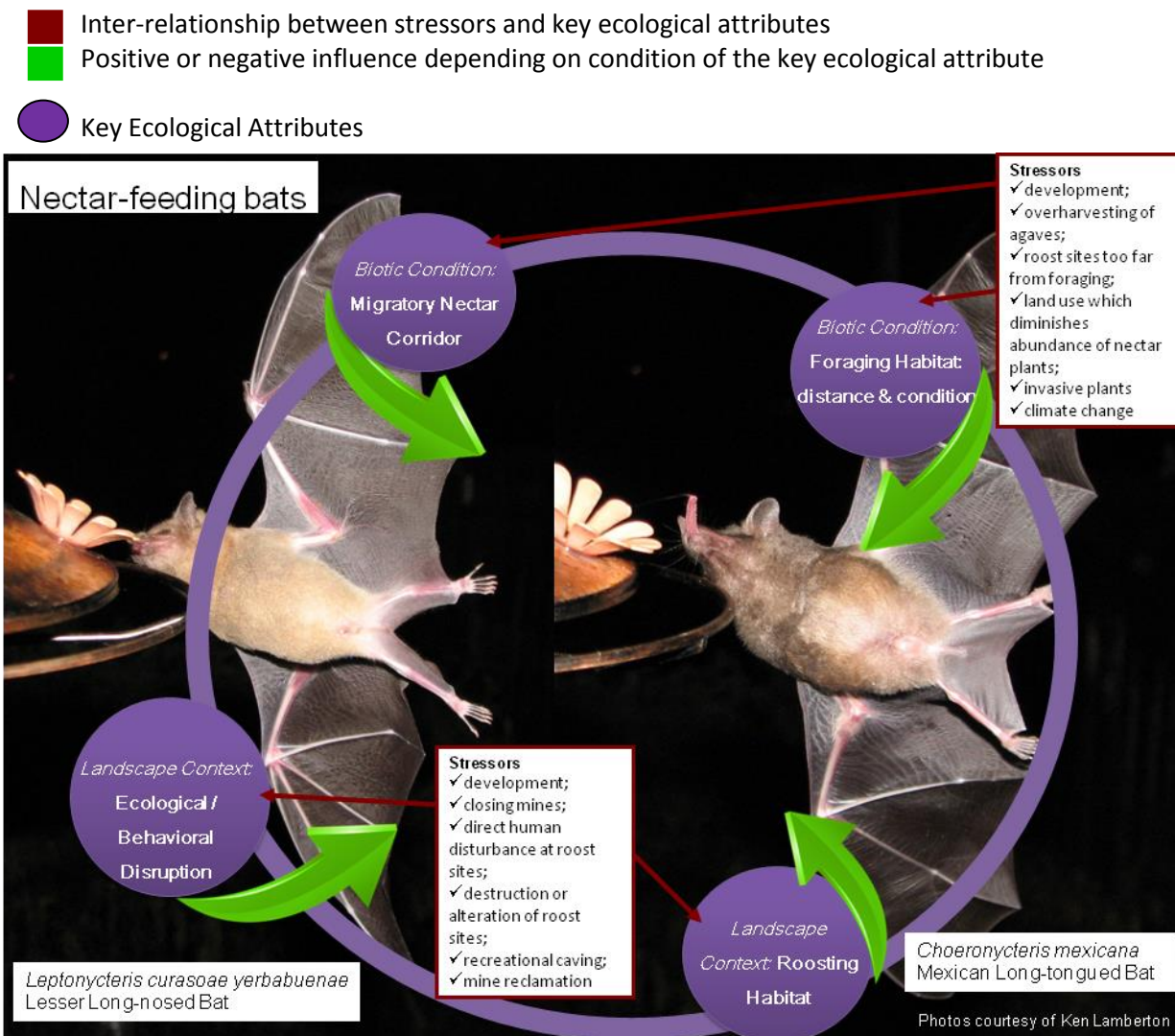
Table D-10. Key Ecological Attributes (KEA) used to determine the ecological status of nectar feeding bats: Lesser long-nosed bat (*Leptonycteris yerbabuenae*), Mexican long-nosed bat (*Leptonycteris nivalis*), and Mexican long-tongued bat (*Choeronycteris mexicana*) occurring in the Madrean Archipelago ecoregion.

KEA Class: Name	Definition	Rationale	Stressors
Landscape Context: Roosting Habitat	The presence of caves, inactive mines, rock crevices and other suitable landscape features for bat roosting.	Bats are very sensitive to human disturbance at roosting sites. They may have a difficult time finding alternative roosts that meet their requirements. Loss of roosting sites is one of the key factors cited in the population decline of nectar feeding bats (AZGFD 2006; 20011, NMGF 1994, USFWS 2007; 1994). Both daytime and nighttime roost sites from which bats can reach foraging habitat are key to species success in the Madrean ecoregion.	Roosting sites are most threatened by human disturbance and by human alteration of these sites such as the closing of mine shafts or improper gating. Roosting sites may also be compromised or destroyed by natural disasters such as cave collapse. Disturbance at maternity roosts can cause bats to drop their pups to flee from the intruder or cause them to abandon their young altogether (Nolan, n.d.).
Landscape Context: Ecological / Behavioral Disruption	The degree to which bats are disturbed by activity in or near roost sites. Mexican long-tongued bats are very sensitive to human disturbance.	Disturbance, vandalism, and outright killing of bats in roosting sites probably have detrimentally affected all three nectar-feeding species (USFWS 1994). Excessive disturbance may lead to increased mortality (e.g., of neonates) and roost abandonment. Activities associated with attempts to control vampire bats in Mexico have resulted in direct mortality and roost abandonment by long-nosed and long-tongued bats, and suitable alternate roost sites for displaced bats may not be available, even if other caves or abandoned mines are present. Bats have low reproductive rates and can recover only slowly from population declines.	Recreational caving, scientific studies in caves or mines, mine reclamation, noise from activity. Disturbance at maternity roosts can cause bats to drop their pups to flee from the intruder or cause them to abandon their young altogether (Nolan, n.d.).
Biotic Condition: [Distance to & Condition of] Foraging Habitat	The amount of habitat with columnar cacti and agave nectar plants within foraging distance of known bat roost sites.	Nectar/pollen feeding bats make considerable seasonal and nightly movements. Nightly foraging flights may be as much as 40 km from the roost site, and foraging areas are selected based on past and present signs of high resource availability (many cacti or agaves in an area) and utilized over several nights until the pollen and nectar resources are depleted (U.S. Fish and Wildlife Service 2013).	Destruction of natural habitats and feeding areas due to human development, human commercial use of agave and browsing of agave blooms. Alterations in the floristic composition due to disturbance (e.g., fire, grazing, agave harvest for tequila)
Biotic Condition: Migratory Nectar Corridors	The nectar corridor that extends from southern Mexico north to southwestern United States.	In their northward spring migration from Mexico, bats travel along defined corridors - Coastal Lowland Route and Inland Montane Route. During migration they rely on foraging from patches of nectar producing plants to refuel for each subsequent leg of the journey (Arizona-Sonora Desert Museum 2013).	Destruction or modification of habitat supporting nectar plants that bats rely on during migration that would disrupt migratory corridors.

D-8.6 Conceptual Model Diagram

A conceptual model diagram for each CE provides a visual summary representation of the Key Ecological Attributes and species' life cycle (Figure D-15). These diagrams are intended to show how various stressors interact with categories of Key Ecological Attributes to highlight important drivers for species ecological integrity. The arrows indicate relationships between stressors and but do not indicate the nature of the influence (i.e. positive or negative).

Figure D-15. Conceptual model for nectar feeding bats, showing key ecological attributes (by class) for these species, and stressors on the KEAs.



D-8.7 References for the CE

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